Understanding Variation in Water Quality using a Riverscape Perspective

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ABSTRACT

With the increasing degradation of rivers worldwide, an understanding of spatial and temporal patterns in freshwater quality is important. Water quality is highly variable in space and time, yet this is largely overlooked at the scale of stream catchments. I employed a landscape ecology approach to examine the spatial patterning of water quality in complex, impacted stream networks on the Canterbury Plains of the South Island of New Zealand, with the goal of understanding how land-use effects proliferate through stream systems.

In particular, I used "snapshot" sampling events in conjunction with spatial modelling and longitudinal profiles to investigate the ways in which spatial and environmental factors influence the variability of water quality in stream networks. Spatial eigenfunction analyses showed that distance measures, which took into account variable connectivity by flow and distance along the stream between sites, explained more spatial variance in water quality than traditional distance metrics. Small upstream reaches were more spatially and temporally variable than main stems (under summer base-flow conditions). The extent of spatial variation in water quality differed between stream networks, potentially depending on linkages to groundwater and the surrounding landscape. My results indicated that the water quality of headwater streams can have a disproportionate influence over water quality throughout an entire network.

I investigated spatio-temporal patterns in water quality more intensively in one stream network, the Cam River, in which I found consistent spatial pattern through time. The relative balance between nutrient inputs (pollution and groundwater) and in-stream conditions influenced the spatial pattern of water quality, as well as that of several ecosystem processes which I measured simultaneously. The spatially intensive and explicit approach has allowed identification of key factors controlling water quality and ecosystem processes throughout the Cam River. This research highlights the importance of taking a spatially explicit approach when studying stream water quality and that such an approach could be insightful and will contribute to solving current stream management problems.

CHAPTER ONE - UNDERSTANDING VARIATION IN WATER QUALITY USING A RIVERSCAPE PERSPECTIVE

An example of the heterogeneity of stream conditions within a single, relatively small catchment, the L2 River on the Canterbury Plains.

Early stream ecologists identified that streams are the product of their catchments (Hynes 1975). However, that streams are connected dendritic networks, has only recently been included explicitly in stream research (Stewart-Koster et al. 2007, Brown and Swan 2010). The flowing, branched nature of streams means that water quality within a stream network is intrinsically spatially correlated (Tu and Xia 2008). However, development of methods to describe spatial patterns within stream systems has lagged behind those in terrestrial ecosystems (Blanchet et al. 2008b). In this thesis I aimed to show, not only that taking a spatially explicit approach was necessary when studying stream water quality, but also that such an approach could be insightful and will contribute to solving current stream management problems.

1.1 DEVELOPMENT OF THE RIVERSCAPE APPROACH

Spatial heterogeneity is the most salient feature of landscapes (Wu and Hobbs 2007). Understanding the patterns, causes and consequences of spatial heterogeneity for ecosystem function is a key research topic in many branches of ecology (Dutilleul and Legendre 1993, Turner and Cardille 2007). Landscape ecology theory holds that heterogeneous spatial patterns matter, as they set the context for organic processes such as fluxes of organisms, matter and energy. Landscape ecology sees various scales, and the interplay between these, as important (Fausch et al. 2002, Talley 2007). This approach has traditionally been carried out in spite of the statistical noise from spatial variation. However over the last decade, landscape ecology has made great advances in theory and practice (Wu and Hobbs 2007). What has been described as the “emerging science of scale” (Wiens 2002) has led to the recognition that studying spatial structure is both a requirement for ecologists and a challenge. This is a new paradigm (Legendre 1993). Some idea of the emerging importance of space can be gleaned from the bibliometric approach. For example, a clear increase in the number of publications using the words “spatial” and “autocorrelation” in the title or abstract can be seen after 1990 (Rangel et al. 2006). Many

new statistical packages are available which incorporate new statistical methodology developed to treat space more explicitly in analyses (Rangel et al. 2010).

Streams are an integrating element in complex landscapes incorporating much spatial and temporal heterogeneity (Thompson and Lake 2009, Brown and Swan 2010). Many landscape theories have been applied, tested and developed for streams (Thompson and Lake 2009). Drawing on these ideas and incorporating them into freshwater research will allow a more accurate picture of ecological processes to be gained (Fausch et al. 2002). Early models of stream ecosystems, such as the River Continuum Concept (RCC), incorporated landscape elements only in a basic way. The RCC considered physical, chemical and biological attributes of a stream to be changing along longitudinal gradients from headwaters to lower reaches (Vannote et al. 1980). Gradually the consideration of landscape ideas has moved into stream ecology. The serial discontinuity concept made some headway towards a concept of rivers as interrupted continua (resulting from human alteration). This concept also included the idea that a change in any physical or biological parameter may be felt in an upstream or downstream direction of a flow obstruction (Ward and Stanford, 1995). The effects of disturbance from water abstraction, land-use changes and point source discharges can similarly act in both directions (Pringle, 1997). Recently a multidimensional perspective of rivers has been developed. Vertical and lateral linkage between river channels and the adjacent subsystems are important; hyporheic, parafluvial and riparian, have now been included in various models (Fisher et al. 1998, Junk and Wantzen, 2004).

In recent years stream ecology has emphasised links between streams and landscapes. Terrestrial frameworks such as patches, fragmentation and hierarchies have moved into the freshwater realm (Boulton et al. 1997, Torgersen et al. 1999, Talley 2007). In the previously mentioned concepts it is recognised that rivers are influenced by the landscape through which they flow. However, the spatial scale under which these concepts are often applied is small. Although stream systems have long been recognised as having hierarchical structure, most

studies still take place at the reach scale, which itself is poorly defined. However, stream management must occur at larger scales, within catchments, or sub-catchments (Lowe et al. 2006, Thompson and Lake 2009).

Various attempts have been made to embed these ideas in a larger scale approach (Melbourne and Chesson 2006) and studies have been conducted incorporating whole catchments, referred to as the "riverscape" perspective (Allan 2004). From this standpoint, rivers are investigated as whole ecosystems, strongly influenced by their surrounding landscapes, and at multiple scales (Wiens 2002, Allan 2004). The wider landform through which a stream flows varies in terms of geology, hydrology, vegetation, topography and climate. Landscape factors exert variable influences over in-stream parameters, processes and biota throughout systems. The importance of considering how the configuration of tributaries and main channels within a stream system relate to the wider landscape has been recognised (Townsend et al. 2003, Allan 2004). However, many studies do not account for the reality of the connected configuration of stream networks. This is of particular consequence in riverine networks, which consist of linearly arranged, hierarchical, branching habitats (Fagan 2002, Brown and Swan 2010). Fagan (2002) predicted that because of their hierarchical nature, the within-network flux of organisms would be greater through mainstems relative to headwaters, as these include movement of organisms and matter from and between branches. The dispersal rate of organisms and matter are disproportionately important in riverine systems due to the asymmetric movement of energy and matter from headwaters to stems. Consequently forces structuring communities may vary drastically in different parts of a network (Brown and Swan 2010). Researchers who study water quality in stream networks encounter issues that go beyond those already mentioned. Dendritic landscapes have a striking mismatch between the geometry of dispersal, "as the trout swims", and the geometry of disturbance, "as the crow flies" (Fagan 2002). Moreover, it has been suggested that stream distance is a more appropriate representation than Euclidean distance,

when the travel path between two locations is restricted to the stream (Fagan 2002, Gardner et al. 2003, Lyon et al. 2008).

The source of water in streams is spatially variable, with the intermixing of ground and surface waters occurring at different spatial scales and dimensions (Gregory et al. 1991, Cruezé des Châtelliers et al. 1994, Dahm et al. 1998). Landscape elements, such as tributaries, channel gradient, width to depth ratio, sinuosity, relic channels, geolithology, channel morphology and stream bed composition, add further complexity to the exchange of surface and ground waters (Dahm et al. 1998, Wayland et al. 2003). These may cause variation in the intermixing of surface and ground waters, which in turn may cause spatially variable water quality. Artificial channelisation and straightening speed the flow of surface waters and minimise connectivity to ground water and riparian systems. Such reaches are thought to function more like a pipe with little nutrient and sediment retention (Dahm et al. 1998). Alternatively, tributaries and aggrading reaches have increased interaction with ground water (Gregory et al. 1991). Spatially heterogeneous geomorphic processes are discontinuous due to variation in grain size, tributary inputs and vertical segregation through variable aggradations and degradation (Cruezé des Châtelliers et al. 1994). However, most water quality research is focused at either the smallest or largest scales (Dahm et al. 1998, Lyon et al. 2008). Fine scale studies include those of the path of water into the stream at single location (Peterjohn and Correll 1984, Sheridan et al. 1999) and reach-scale research on nutrient cycling (Niyogi et al. 2004). At the other end of the spectrum, catchment water quality is usually monitored on a regular basis at a small number of locations in a catchment, generally focused at the catchment outlet. This integrates the effect of all the point and non-point source processes occurring throughout the catchment. However, effective catchment management requires data which identifies major sources and processes which operate between these scales (Grayson et al. 1997, Salvia et al. 1999).

Classic theory on the processes influencing catchment water quality comes from two sources, catchment ecology and nutrient spiralling. Research in the Hubbard Brook catchment

involved nutrient budgets of precipitation inputs and stream outputs; this typified the catchment approach (Lowe and Likens 2005). In the catchment approach, riparian and hyporheic zones held little importance, but chemical changes of water moving through these zones suggest they play an important role in nutrient transformation (Peterjohn and Correll 1984, Valett et al. 1996). From the nutrient spiralling perspective, nutrients were studied in a production sense, where the focus was on budgets within the wetted stream only (Newbold et al. 1982, Grimm 1987). Nutrient cycling processes have been shown to change the concentration of nutrients in the water column, but the focus is turning to where these processes occur and their relative importance (Finlay et al. 2011). Merging these perspectives will increase our understanding of catchment nutrient processes (Dahm et al. 1998). To be meaningful, studies considering the outflow of solutes from a drainage basin must also consider the aggregation of upstream solute responses and channel routing (Walling and Webb 1980). Typically studies lump the whole drainage basin together in analysis, not considering basin size or heterogeneity. More particularly they mostly ignore the fact the upstream drainage basins may be additive in their downstream effect, and that the course of this effect is determined by channel routing (Walling and Webb 1980). Knowledge of hydrological flow paths is critical in linking the hierarchy of stream networks (Dahm et al. 1998). Thus, in Chapter Two, I examined spatial patterns in water quality within five complex dendritic stream systems with a focus on factors influencing the level of spatial variability observed in each system.

1.2 BASE FLOW WATER QUALITY SAMPLING

Besides spatial autocorrelation, variation in flow levels, not only during flood events; influences the concentration of many nutrients and levels of physico-chemical parameters (Meybeck et al. 1999, Scarsbrook 2002, Wayland et al. 2003). The water chemistry of rivers during base flow is therefore more likely to represent the catchment geology and land-use than during flood flows or droughts (Wayland et al. 2003). Thus, base flow sampling avoids the bias and influence of rivers

being sampled in particular states of flow, which would only further complicate analyses (Biggs et al. 1990). Pionke et al. (1996) define base flow as the period between storms, when the hydrograph is in the later stages of the recession limb, usually, more than two days post storm flow peak.

Early base flow surveys were conducted from the perspective that in low flow, when most flow comes from subsurface water stored in soil and rock, one would expect that water quality will vary according to the geology within small catchments (Walling and Webb 1975). It has been suggested that under base flow conditions, surface runoff is limited, so water is heavily influenced by point sources and ground water, thus base flow conditions are a good time to identify these effects (Eyre and Pepperell 1999). On the other hand, the continual presence of base flow means that water chemistry in this period represents an integrated signal of climate, geology, historical and present land-use throughout the watershed and should help link water quality to land-use distributions (Wayland et al. 2003). The mean transit time of water (thus, contact with the landscape) is also highest at low flow, so under these conditions, it is thought that water quality will be tightly coupled with land-use (Lyon et al. 2008).

Base-flow conditions can exist for long periods, for example 10-11 months per year in New Zealand (Biggs et al. 1990), so water quality at these flow levels is an important constraint on the health of in-stream biological communities (Grayson et al. 1997, Lyon et al. 2008) and is critical for water allocation (Biggs et al. 1990). However, caution should be used in interpretation of data from only one sampling occasion, as base flow water chemistry may be temporally variable in some watersheds (Clow et al. 1996) yet stable in others (Pionke et al. 1999). As a large portion of nitrates, phosphates and sediments are exported during floods, one cannot scale-up to yearly catchment exports using base flow measurements (Dahm et al. 1998, Eyre and Pepperell 1999, McKee et al. 2001).

1.3 THE "SNAPSHOT" APPROACH

Heterogeneity in both landscape and water quality exists within single catchments, even those that are small (Prowse 1984) or in pristine condition (Clow et al. 1996, Finlay et al. 2011). The consideration of sites across a network is important for improving understanding of the fate and transport of nutrients, as well as for the identification of causal relationships (Tu and Xia 2008, Finlay et al. 2011). A spatially intensive approach to water quality monitoring, which involves the collection of water quality data from a large number of sample sites over a short period of time, will achieve the best understanding of the processes occuring (Eyre and Pepperell 1999). Walling and Webb (1975) were one of the first to publish a spatially intensive water quality study. This involved relating conductivity within a catchment to geology and land-use. The spatially intensive approach was not used significantly, however, until the 1990's. It has been redefined as "snapshot" or "synoptic" sampling (Grayson et al. 1997, Lyon et al. 2008 respectively).

The "philosophy" of the snapshot methodology is to sample a river at every confluence point and discharge point at an instance in time (Walling and Webb 1975, Grayson et al. 1997). The instant can be considered a period in which all parts of the river are at constant flow. As most water quality parameters vary with discharge and floods, sampling during changing flow would contain discharge-related variation, making spatial analysis difficult (Grayson et al. 1997). Sampling in this way provides insight into the biogeochemical behaviour throughout a stream network at low flow conditions and is in tune with emergent paradigms relating to spatial and network perspectives in fresh water ecology (Fausch et al. 2002, Lyon et al. 2008). Lyon et al. (2008) described a synoptic sampling campaign, with 100 or more sample locations, as still spatially sparse compared to the heterogeneity found in natural stream systems. However, published studies of snapshot sampling campaigns range between 50 sites sampled over 12 hours (Salvia et al. 1999) to 108 in 3 days (Eyre and Pepperell 1999), with the most intensive being over the longest time frame; over 500 in 12 days (Walling and Webb 1975). An exceptional

study is that of Dent and Grimm (1999), who collected around 40 samples almost simultaneously along a 10km stretch.

The major advantages of the snapshot method relate to the bulk of additional information that can be gained relatively quickly, compared with repeated data collection at catchment outflows. This method has enabled insights into system behaviour, including quantification of unknown point discharges, identification of key in-stream sources of suspended material and the extent to which biological activity (phytoplankton growth) affects water quality (Grayson et al. 1997). The snapshot methodology also allows the mass balance approach to be used. Using this approach allows all point-source total loads (tributary and drain inputs in this case) to be subtracted from the total loads at main stem sites in a longitudinal manner (Grayson et al. 1997, Salvia et al. 1999, Behrendt and Opitz 1999). In this way, snapshot sampling has been used as an independent check of licensed discharges (Grayson et al. 1997). The large data sets provide opportunities for cross-correlation with parameters that change across catchments. They have allowed identification of geology, land-use and point source contributions to water quality, and management action against point source contribution to be taken (Eyre and Pepperell 1999). Snapshot sampling is often used to complement multiple measurements taken through time at one site (Grayson et al. 1997). The regime sampled must be representative of base flow conditions, as sampling in this way during a flood event would not meet the snapshot criteria (Eyre and Pepperell 1999). Most studies which took a snapshot approach in sampling water quality concluded that this method has been under-utilised and that it allowed valuable insight into the properties of the system studied.

Repeated snapshot sampling events within a stream system have the potential to help explain the level of temporal variability in stream water quality and have been associated with flow, as well as changes in the spatial pattern of water quality through time (Clow et al. 1996, Scarsbrook 2002, Wayland et al. 2003). The way streams interact with landscape factors through time can vary, and the source of water may change (Dahm et al. 1998, Wayland et al. 2003). This

may result in temporal changes in spatial water quality patterns, yet only a handful of snapshot studies have been conducted on a repeated basis. The logistical difficulties and cost involved in snapshot sampling mean such studies have repeated the sampling event only three times across variable time frames and successional stages of the hydrograph (Clow et al. 1996, Dent and Grimm 1999, Wayland et al. 2003,). McKee et al. (2001) used a combined approach, sampling using snapshot methodology six times at up to 79 sites across all seasons, routinely once a month at subcatchment outlets and also event sampling up to six times a day during high flow events. This sampling regime allowed patterns in water quality, longitudinally between sub-catchments and through time, to be identified in this system (McKee et al. 2001). However, long term spatial studies are rare. Exceptions are Prowse et al. (1984), who sampled 25 points many times over three years, and a long-term monitoring project on the Seine River in France, where 236 stations were spread across orders 1-8 and were sampled over a 30 year period (Meybeck et al. 1999). Spatial changes in the physico-chemical composition of stream water through time may indicate that the influence of various landscape and in-stream factors controlling water quality has changed.

Short term variation in antecedent moisture, soil-water, microbial and dilution processes will also vary (Pionke et al. 1999 and 1996). Thus, even base-flow water chemistry may be temporally variable in some watersheds (Clow et al. 1996). Short term variations in water quality can have flow-on effects in governing fish and insect assemblages and population distributions (Stewart-Koster et al. 2007, Brown and Swan 2010). Such variation may be disproportionately important under base-flow conditions, which can persist for much of the year. In light of the lack of consensus in literature surrounding the role of slight variations in flow in controlling spatial patterns in stream water chemistry through time, in Chapter Three I investigated spatio-temporal variation in base-flow water quality in one of the five stream networks that I studied in Chapter Two, the Cam River. In reducing the scale of study to a single catchment, I was able to sample in

a more spatially intensive and repeated manner, by undertaking multiple snapshot sampling events.

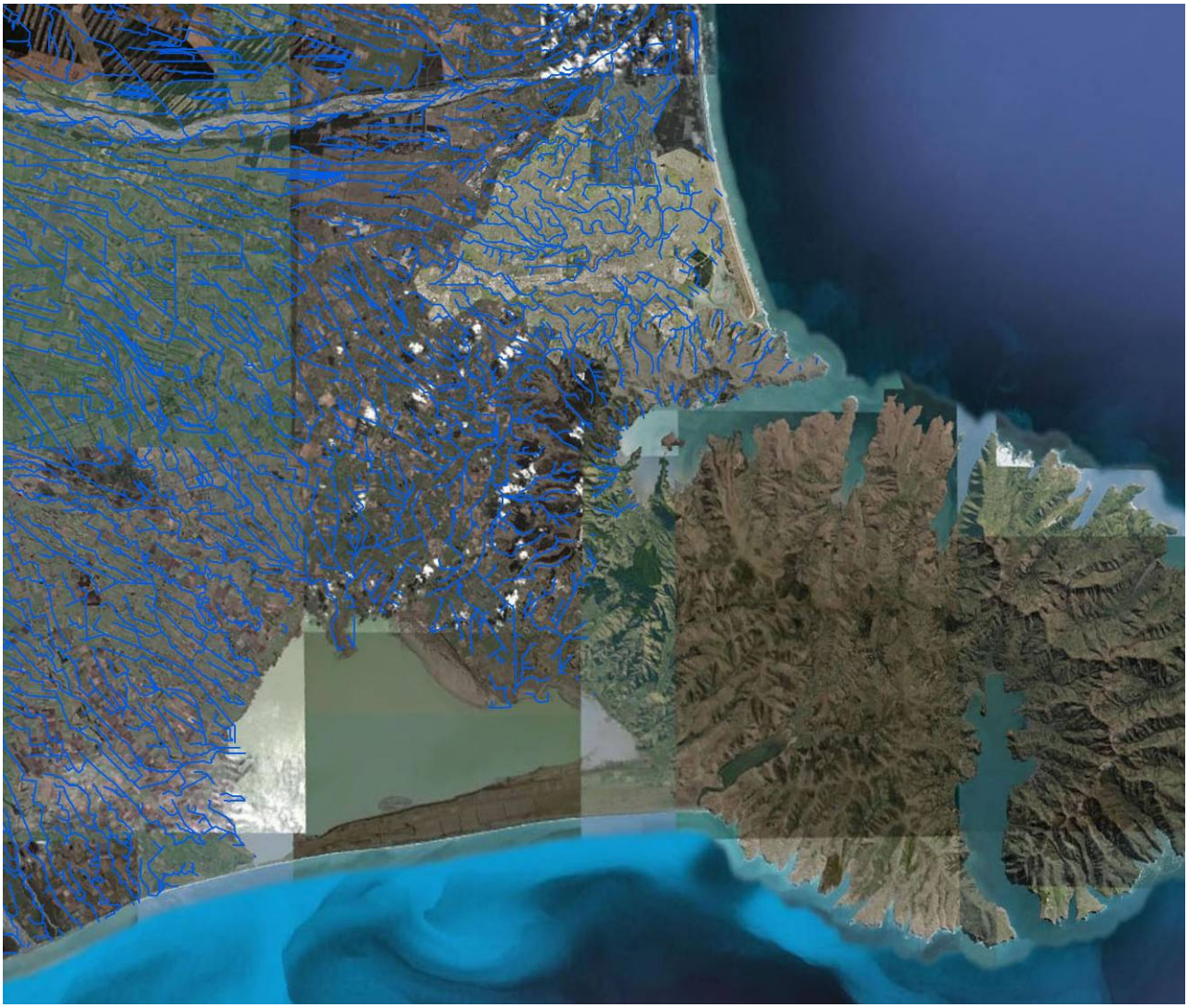
Stream ecosystem processes, such as primary productivity and the turnover of organic matter respond to the range of conditions experienced through time, in this way integrating some of the temporal water quality variability often experienced (Biggs and Kilroy 2004). Such processes are useful as a surrogate for water quality measurements, which are more expensive (Bott et al. 1985). Ecosystem processes have direct impacts on consumers, the invertebrates and fish which inhabit streams; and thus are of vital importance to sustaining aquatic biodiversity (Winterbourn 2004). When measured through time (as bioaccumulation or breakdown rates) at multiple points in a stream network, ecosystem processes may provide a "snapshot" of vigour and resilience of biotic communities throughout.

The drivers and stressors of ecosystem processes have been studied at the large scale, comparing processes across biomes (Bott et al. 1985), regions (Mullholand et al. 1987, Findlay and Sinsabaugh 2006), stream types (Biggs and Close 1989) and between catchments (Quinn et al. 1997). Comparisons to reference sites (Carlisle and Clements 2005, Ferreira et al. 2006) and artificial enrichments (Gulis and Suberkropp 2003) are also common in stream productivity research. Worldwide, studies of within-catchment variation in ecosystem processes are rare, and the few I have found took place in relatively pristine catchments (Clow et al. 1996, Finlay et al. 2011). The lack of research into (and consequently knowledge of) within-catchment variation in ecosystem processes may come from the fact that we know a lot about a few headwaters – such as Hubbard Brook, yet processes in these systems cannot be applied to all (Bishop et al. 2008). The study of degraded catchments, such as the Cam River, provides the opportunity to identify a variety of relationships, which may be applicable to other complex, impacted systems. In Chapter Four of this thesis I studied spatial variation in various ecosystem processes in the Cam River, (which I had measured simultaneously with the snapshot water quality sampling events). In this Chapter, I used spatial-environmental variance partition methods (first developed for

community ecology), in a novel way, to tease apart influences of anthropogenic impacts and spatial location on variation in ecosystem processes within the Cam River system.

This thesis is structured as a series of stand alone papers intended for development towards publication in international scientific journals. This means there is necessarily some repetition, particularly in the Methods sections, but this is an effective and efficient way to present the multiple aspects of this work. All analyses and writing are primarily my own, with contributions of co-authors listed in the acknowledgements. Throughout the thesis, chapters are referenced by chapter number as they appear in this thesis. Figures and tables are numbered within each chapter, however the complete reference list is provided for all chapters at the end of the thesis.

CHAPTER TWO - SYSTEM-SPECIFIC SPATIAL VARIABILITY OF STREAM WATER PHYSICOCHEMISTRY



The myriad of spring, lowland and foot-hills fed channels, as well as water-races and irrigation canals, that cross the Canterbury Plains, forming many complex stream systems (as defined by the River Environment Classification (REC, Snelder et al. 2005) (image from Google Inc. 2009, combined with the REC in ArcMap 9.2).

2.1 INTRODUCTION

Most of New Zealand was originally covered in forest or indigenous grasslands , yet since human colonisation nearly two thirds of the forest cover has been removed. Conversion of this land to agriculture and increased urbanisation have led to increases in nutrient levels (Houlahan and Findlay 2004, Simon et al. 2007), sedimentation (Suttle et al. 2004; Matthaei et al. 2006), microbial contamination, and temperature (Larned et al. 2004) in freshwater streams, lakes, reservoirs and wetlands and has led to the degradation of New Zealand's inland waters (Parkyn et al. 2003). These influences are linked to the large-scale application of nutrient-rich fertiliser and livestock effluent. The associated loss of riparian vegetation has contributed to these impacts and caused changes in water quality by increasing the connectivity between the stream and the surrounding landscape (Tabacchi et al. 1998; Parkyn et al. 2003). Animals crossing streams and stock pugging have also had a deleterious effect on bank structure, decreasing nutrient retention and increasing sedimentation (Houlahan and Findlay 2004; Collins et al. 2007). These physical changes can impact on the biotic community and ecosystem processes of streams (Kronvang et al. 2005) and can result in the degradation of water quality in streams and rivers.

Lowland streams on the Canterbury Plains exist in a landscape that has become a complex mosaic with urban sprawl and changes in farming practices (Winterbourn 2008). Lowland streams are also complex in terms of dendricity and hydrologic connectivity due to inputs from springs and connectivity with stock and irrigation raceways. This is especially true for first order streams, which make up a majority of the stream network. Small streams have been proposed as instrumental in conditioning the water for export, dampening flood waters, cycling nutrients (Bernhardt et al. 2003) and buffering pollutants (Klaminder et al. 2006); and also for habitat and refugia (Lacey et al. 2007). Due to their high edge to volume ratio and high contact time of water in these areas, processes in small streams are more sensitive to disturbance, including land use and climate change (Bishop et al.

2008; Lowe and Likens 2005). Therefore, knowledge of the small streams that make up the complex river systems on the Canterbury Plains could make a tangible difference to management of water quality as a whole.

The theory of landscape ecology holds that heterogeneous spatial patterns matter, as they set the context for organic processes such as fluxes of organisms, matter and energy (Fortin and Dale 2005; Wu and Hobbs 2007). Early models of stream ecosystems such as the River Continuum Concept (RCC) were limited in their spatial consideration (Vannote et al. 1980). The RCC considered physical, chemical and biological attributes of a stream to be changing along longitudinal gradients from headwaters to lower reaches (Vannote et al. 1980). Gradually, however, landscape ecology ideas have moved into stream ecology (Lowe et al. 2006; Thompson and Lake 2009). The serial discontinuity concept made some headway towards a concept of rivers as interrupted continua (resulting from human alteration) (Ward and Stanford 1995). Recently multidimensional perspectives of rivers have been developed and a more continuous view, concerning vertical and lateral linkage between river channels and the adjacent subsystems (hyporheic, parafluvial and riparian) along the river continuum, have now been included in various models (Fisher et al. 1998; Fausch et al. 2002; Junk and Wantzen, 2004).

In this study, I employed a landscape ecology approach to examine the spatial patterning of water quality in stream networks, with the goal of understanding how land-use effects proliferate through complex river systems. I examined the spatial variability in physico-chemical water quality within five stream networks on the Canterbury Plains of New Zealand. I first investigated how stream size and position with a network affected physico-chemical water quality. I hypothesised that streams of different size would exhibit a different range of water quality conditions. I went on to test the hypothesis that the overall spatial patterns in water quality differed between systems. Spatial patterns were assessed by examining the level of spatial similarity that existed between sites within

each system. Various distance measures and system level properties were used to identify factors that may govern spatial patterns.

2.2 METHODS

2.2.1 Site Selection and System Mapping

Canterbury has over 27 000 km of running river channels, more than any other region in New Zealand (Winterbourn 2008). Five predominantly spring-fed river systems were selected in the East Coast Plains Ecoregion of the Canterbury plains, which extends from the foothills of the Southern Alps to the East Coast, excluding Banks Peninsula (as recognised by Harding and Winterbourn 1995) (Figure 1). The rivers are situated on the peripheries of alluvial fans produced by the Waimakariri and Selwyn Rivers (Winterbourn 2008). All five systems are considered lowland fed rivers, a group that typically have spring sources and little hydrological variability. Subsurface seepage can provide a substantial proportion of flow and heavy rainfall results in freshes, but these are rarely major (Biggs 1985, Winterbourn 2008). Soil patterns on the plains can be deduced from geomorphic history and are more complex than previously thought. Remnants of old alluvial fans are sometimes preserved in association with small rivers (Webb 2008). Canterbury's weather is affected by oceanic and mountain effects with precipitation spread approximately evenly across the plains, 600-800mm annually, with no strong seasonal variation (Sturman 2008). Riparian species naturally occurring are flax, toetoe, grasses and sedges (Winterbourn 2008). Plains streams vary in degree of naturalness, from stony-bottomed to drains that are channelised and often carry water from elsewhere on the plains (Winterbourn 2008). Some streams are abundant in periphyton and rooted macrophytes, which are home to a specific set of invertebrates and are periodically cleared by councils and farmers (Winterbourn 2008). Since settlement in the 1850's (Hawkins 1957), the

Canterbury Plains have developed into what is now a patchwork of many types of agriculture, lifestyle blocks and small towns. The systems I studied were selected with a preference towards more complex dendricity, for the potential to reveal interesting spatial patterns. Three of the five streams pass through small towns, and the Styx River through parts of Christchurch (Figure 1).

All waterways flowing into each of the five systems were mapped; this included all drains, stock and irrigation raceways that joined the main stem. The flow path of water on the plains often differs from that delineated on both topographical maps and the River Environments Classification (REC), a GIS-derived database of network topology of New Zealand's rivers and streams (Snelder et al. 2005), due to the low elevation and complex alterations. Thus, I built a map of each river by tracing the river lines using Google earth images (the most accurate and high quality images available), in combination with field knowledge and ground truthing, using a GPS (Garmin GP560) (Google Inc. 2009, GIS river lines created in ArcMap 9.2).

Sites were placed throughout each system as per the "snapshot" methodology, in order to produce an instantaneous picture of all concentrations and fluxes in a watershed by sampling every confluence and discharge point within a time period that is as short as possible and where flow is stable (Salvia et al. 1999, Wayland et al. 2003, Walling and Webb 1975). Sites were placed where feasible on all first order tributaries and point sources entering the system (sources with flow less than 0.0005 m^3s^{-1} were not sampled). They were situated at least 25 m above and below every point of confluence, so that no two sites were less than 50 m apart. Sites were also placed along uninterrupted main stem and tributary reaches, approximately 500 m apart (Figure 1). This method of site selection resulted in systems having between 21 and 47 sites (Figure 1).

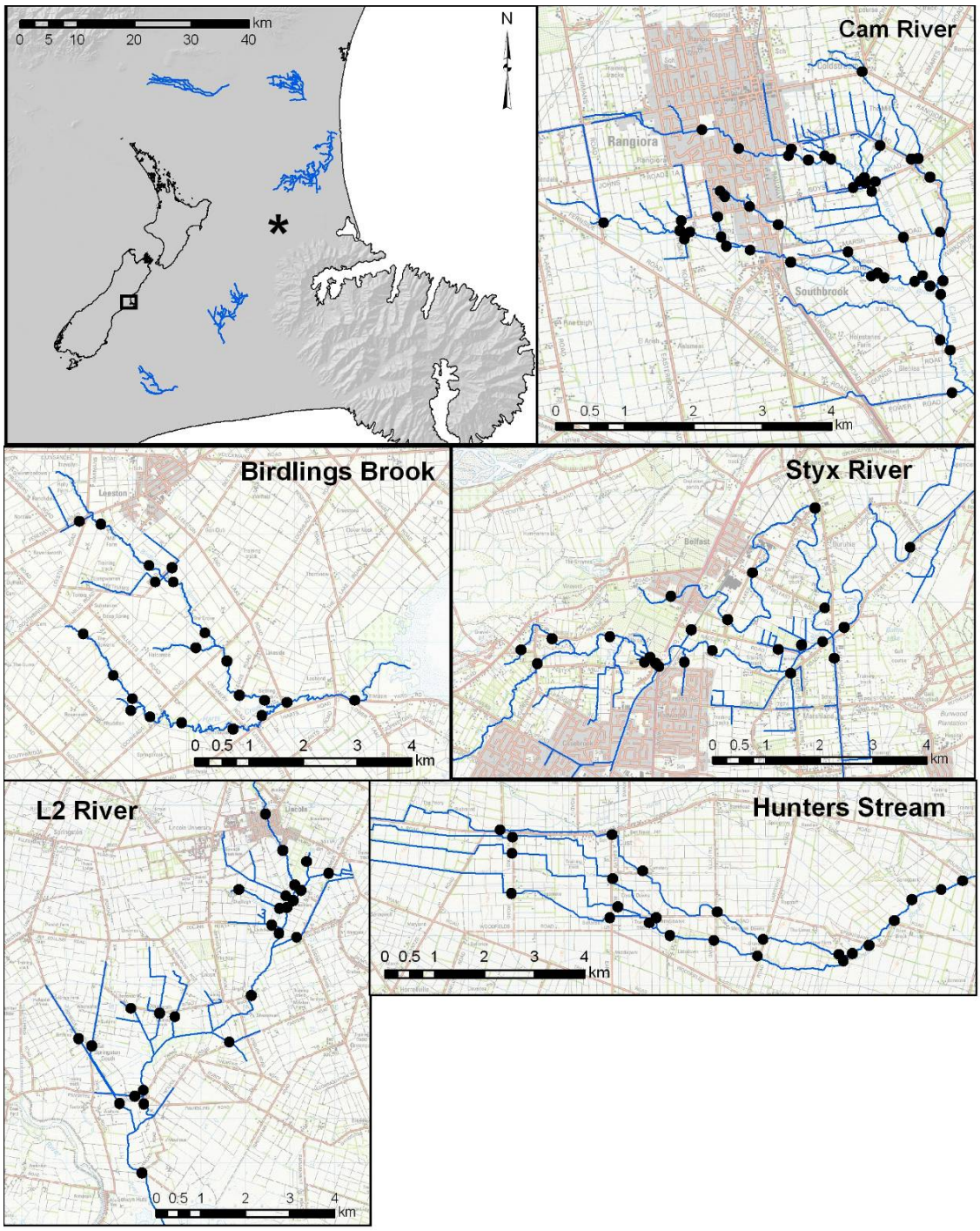


Figure 1: The five stream networks studied were located on the Canterbury Plains of the South Island, New Zealand. Shading in the upper left panel represents the topography of the area, the star is the location of central Christchurch and the north arrow applies to all maps. The lower panels show the natural stream channels of five networks at and the dots show the locations of sampling sites. (Note the different scales in each panel). River lines were constructed by tracing from Google Earth images and ground truthing (Google Inc. 2009). The river lines and sampling sites are overlain on a topographical map of the area, on which urban areas are shown by darker shading.

2.2.2 Physico-chemical water quality sampling

Sites were sampled between March and April 2009, during base flow and were visited on a stream by stream basis, sampling sites close together along a network on the same day so as to minimise temporal variation in the data from each system. Base flow was defined as the period between storms when the hydrograph is in the later stages of the recession limb usually 2 days post storm flow peak (Pionke et al. 1999). Thus, sampling was not conducted if rain had occurred in the previous 3 days in the catchment, Southern Alps or foothills, which may cause the Waimakariri, Ashley and Selwyn rivers to rise, raising the water table feeding springs that contribute to several of the systems. Each day I visited downstream sites first and collected water samples immediately on arrival at a site to avoid contamination from sediments disturbed by entering the water. Sampling was conducted up to 5 hours either side of midday in order to minimise diurnal variation in parameters such as dissolved oxygen and temperature. Diurnal variation in water chemistry is minimal with regards to all major forms of nitrogen and phosphorus except for ammonia (not analysed here) (Finlay et al. 2011). In keeping with the aforementioned criteria, sites within each system were sampled over a period of 3 to 10 days. To understand variation in water quality throughout entire stream systems, one must sample in a spatially intensive way. In doing so, I accept that my study could only be a "snapshot" in time, as a large-scale, spatially intensive study needs many sites to be effective and my sites could only be sampled once.

At each site a 20-m reach was chosen to represent the general condition of the wider reach, measured along the thalweg of the stream channel. I measured 8 physical and chemical properties of the water at each site, using a YSI Sonde 6600, a multi-parameter water quality meter to continuously log information at each site and taking water samples on ice for laboratory analysis. Discharge and wetted width were measured at 3 points along the reach. Due to the large number of sites sampled, a semi-qualitative approach was chosen to assess the physical habitat at each site

(Harding et al. 2009). Physical in-stream properties assessed throughout the 20-m reach were algal and macrophyte cover and a substrate index. An assessment of riparian conditions was also undertaken at each site. Detailed field and laboratory methodology is provided in Table 1.

Table 1: Field and laboratory methodology for the eight water quality variables and other predictor variables measured at each site, as well as treatment in analyses.

Variable	Units	Field and/or laboratory methodology	Reference	Treatment
Temperature	ºC	Time averaged spot measurement. The YSI Sonde 6600, a multi-parameter, water quality measurement device was placed in the thalweg of a run at the top of each site, continuously logging during time spent at each site. The final 5-20 minutes of data from each site was averaged (visit times varied depending on sample run intensity).		Eight response water quality variables
Dissolved Oxygen	mg/L			
pH				
Conductivity	µS			
Turbidity	NTU			
Dissolved Nitrite-N + Nitrate-N (DIN)	mg/L	I collected 80 mL of water in a syringe from the thalweg of the stream just below the surface and pushed through a filter (Whatman GFF 250 Millipore rating) into a 100mL opaque plastic bottle (pre-soaked in 5% hydrochloric acid overnight then rinsed three times with distilled water and a further three times with milli-q water). 80 mL of unfiltered water was collected in a similar way. Samples were kept on ice and then frozen. Filtered samples were thawed then analysed colourmetrically for DIN (assumed from combined nitrate-nitrite due to low nitrate concentrations).	American Public Health Association (APHA), 1995	
Soluble reactive phosphorus (SRP)	µg/L	SRP using an automated, high throughput, water chemistry machine, the Easychem Plus (Systea, Italy). All chemicals used to make standards and reagents were of reagent grade and methods used by machine standard. The unfiltered water samples were thawed then processed for total phosphorus (TP) using standard colourmetric methods and testing for absorbance on a Trilogy Laboratory Fluorometer using a PO4 module.		
Total phosphorus (TP)	µg/L		Wetzel et al. 2000	
Riparian score		Semi-quantitative protocol using P2d – Riparian procedure.	Harding et al. 2009	Predictor variables
Substrate index	%	A metric based on data gathered using P2c – In-stream habitat procedure.		
Algal cover	%	Visual estimate of % of the stream bed covered by algae.		
Macrophyte cover	%	Visual estimation of % of the stream bed covered by macrophytes.		
Shading	%	Visual estimate of % of the stream bed shaded when the sun is overhead.		
Discharge	m^3/s	Discharge measured across one run, evenly flowing and free of obstructions. Offsets were placed wherever depth or discharge changed noticeably, with no fewer than five per transect. Water depth read on the downstream side of a ruler and water velocity is measured four-tenths of water depth up from the bed using a Marsh-McBirney Flow Mate. Discharge was calculated based on standard methods.	Gordon et al. 2004, Harding et al. 2009	
Wetted Width	m	Wetted width was measured at five locations along the 20m reach.	Harding et al. 2009	

2.2.3 Land use, riparian conditions and other GIS derived variables

For each system, catchment area was defined using the River Environment Classification (REC) (Snelder et al. 2005). Due to the low topography of the Plains, catchments generated from digital elevations only approximate actual catchment areas, so manual changes were made to fit known water movement through ground truthing. This is further complicated by stock water races and irrigation canals, which result in considerable cross-catchment transfer of water. To estimate the relative inputs of these artificial waterways between sites, an upstream distance measure which included their length was calculated, as well as their proportional contribution to the overall discharge from each system. The elevation change in each catchment was also derived from the REC (Snelder et al. 2005), while stream order and the number of junctions were counted manually. All parameters related to stream channel distance are based on the manually constructed stream maps. Measures of network dendricity included drainage and confluence density (Benda et al. 2004). The contribution to each system by ground water was derived as the number of springs and proportion of confined aquifer in the catchment area (Environment Canterbury GIS Layers).The area of each catchment situated within a 100-m buffer of the stream channel was also calculated. For each catchment and 100-m buffer zone metrics of proportional land use, forest presence, urban development and road density were derived.

2.2.4 Distance Measures

Various matrices of the distance between each site and every other site in each network were constructed. The Euclidean pair-wise distance was calculated between the sites in each system from the NZMG (New Zealand Map Grid) coordinates of each site. This was calculated in R using that spatial Statistics add on (spatstat, R Development Core Team 2007) to create a Euclidean distance matrix, representing direct or overland distance between sites. A stream distance matrix was

calculated using the mapped streams and the spatial analyst function to better represent the actual path water takes across the landscape (ArcMap 9.2). To account for variable connectivity, a binary connectivity matrix was constructed for each system, with sites that were connected directly by flow attributed a one and all other sites attributed zero (Dray et al. 2006). For example sites on two separate first order tributaries of the main stem are connected to all sites downstream on the main stem and sites upstream on their individual branches, however not by flow to each other. Finally a stream distance matrix, which was weighted by the inverse of the average flow encountered at sites between any pair of sites, was constructed to better represent the ease of transfer of solutes and particles between sites.

Table 2. List of all system wide metrics and method of derivation, calculated for each of the five systems as a whole.

System wide variables	Methodology/calculation (all variables are based on the stream maps I constructed)	Units
Natural stream length	The length of stream (including all tributaries) in the natural catchment upstream of the lowest site based on traced and ground truthed stream maps. The decision was made to "cut off" a natural stream reach (beginning the "unnatural" portion), when upstream reaches took the form of highly channelised, free-flowing canals, of uniform depth, characteristic of water-race or irrigation networks (Google Inc. 2009, ArcMap 9.2).	m
Length of all waterways entering a system	The length of stream including all natural and man-made waterways upstream of the site. Where branches split in two in the direction of flow, I only measured the branch connecting to the system. (Waimakariri District Council water-race data in geospatial form and ground truthing).	m
Proportion of network that is water-race fed	The combined volume of water entering the system (instantaneous discharge) at points of "cut off" from the natural network, expressed as a proportion of the discharge of the most downstream site.	
Total number of junctions	Counted manually (excluding the water-race network).	
Number of springs in catchment stream	Calculated using the Environment Canterbury online GIS database, springs found to date by Ecan field workers, (Environment Canterbury GIS layers).	
Catchment area	Catchment areas were defined using the River Environments Classification (REC), then altered manually to meet known directions of water flow and extra channels not delineated by this model (Snelder et al. 2005).	km^2
Area of 100-m buffer zone	The area upstream of each site within 100m of the natural stream. This distance was chosen based on its use in buffer zone delineation (Baker et al. 2007) and as the smallest distance class to measure land use metrics to assess critical distances of impact (Houlahan and Findlay, 2004) (ArcMap 9.2).	km^2
Length of road per km^2 of catchment area	The total length of all paved and metalled roads in a catchment area (New Zealand Landcover Database ver.2, Terralink 2004) expressed as a fraction of the area in km^2.	
Proportion catchment area that is urban	The proportion of the catchment area that is built-up (New Zealand Landcover Database ver.2, Terralink 2004))	
Proportion buffer zone that is urban	As above but for the 100-m buffer zone.	
Proportion catchment that is in moderate to high intensity farming	The proportion catchment area that is in use as Dairy, Beef, Sheep or Sheep and Beef (Agribase, 2009)	
Proportion buffer zone that is in moderate to high intensity farming	As above but for the 100-m near zone.	
Proportion catchment that is in dairy farming	The proportion catchment area that is in use as Dairy (Agribase, 2009)	
Proportion buffer zone that is in dairy farming	As above but for the 100-m buffer zone.	
Proportion catchment with urban cover	The proportion of the natural upstream catchment that is built-up (New Zealand Landcover Database ver.2, Terralink 2004))	
Proportion buffer zone with urban cover	As above but for the 100-m buffer zone.	
Elevation change within catchment.	Based on the reach of highest and lowest elevation in all mapped stream reaches that correspond to REC reaches (Snelder et al. 2005).	m
Proportion catchment with forest cover	The proportion of the catchment covered in exotic or native forest (New Zealand Landcover Database ver.2, (Terralink 2004))	
Proportion buffer zone with forest cover	As above but for the 100m buffer zone.	
Drainage density	Length of natural stream channel per km^2 catchment area (Benda et al. 2004)	
Confluence density	The number of confluences/junctions per km^2 catchment area (Benda et al. 2004)	
Proportion of catchment area that on a confined aquifer	Estimated visually from maps of aquifer locations and known catchment areas (Environment Canterbury GIS layers).	

2.2.5 Analyses

2.2.5.1 Variance in physico-chemical variables associated with stream size

An initial overview of patterns was achieved by plotting box and whisker graphs of each physico-chemical variable against stream order. The variability of the physico-chemical data was analysed by plotting the PCA axis one and two scores from an ordination of all physico-chemical variables, against stream wetted width (variables were transformed to meet normality assumptions), centered and standardised before ordination as is usual for environmental data (Clarke and Corley 2006). Quantile regression was performed to fit regression lines that bound the upper and lower five percent of the data in the R package quantreg using the "br" method, which is a variant of the Barrodale and Roberts (1974) simplex algorithm and is suitable for data sets of this size (Koenker 2009). These were tested for significance using the "boot" method to compute standard errors (Koenker 2009).

2.2.5.2 Spatial patterns in water quality

To initially explore system wide patterns in water quality, each parameter was expressed visually using graduated color symbols to represent the value of each site, at its actual location. To investigate spatial patterns in physico-chemical variables I first tested how spatially structured the water quality was within each system and compared across the five systems. Secondly the four different distance metrics were used to tease apart which type of spatial patterns may be acting in each system. Finally, I examined the cause of spatial patterns, comparing the level of spatial structuring in each system to a range of system level properties.

2.2.5.3 Spatial autocorrelation within systems

To examine the extent of spatial structuring within each system I constructed a series of spatial eigenvectors, based on principal coordinates of neighbor matrices (PCNM) and related methodologies. These methods are based on and comparable to the Moran's I statistics, which are the most commonly used statistics for autocorrelation analysis (Rangel et al. 2006). Eigenfunction analysis produces a set of eigenvectors which each represent an orthogonal spatial structure (as they are the product of a symmetric matrix) (Dray et al. 2006). I used eigenfunction-based spatial filtering techniques to evaluate how well different distance measures accounted for variance in water quality and in-stream conditions as a multivariate suite and individually (Rangel et al. 2006, Griffith and Peres-Neto 2006, Blanchet et al. 2008b).

PCNMs code spatial information in a way that allows one to recover various structures over the whole range of scales that the sampling design passes (Borcard and Legendre 2002). The starting point of the PCNM approach is close neighborhood relationships among sites. First a truncated distance matrix is constructed among sites, using a threshold value (defined as the minimum distance that keeps all sites connected, based on a minimum spanning tree algorithm (Laliberté et al. 2008)), under which all distances are kept as measured and above which they are considered "large" and an arbitrary value is replaced. In the second step, principal coordinates analysis is computed on the modified distance matrix, necessary to represent the spatial information in a form compatible with multiple regression and canonical redundancy analysis (RDA) (Borcard and Legendre 2002). One, or several null and several negative eigenvalues are obtained, these cannot be used as they correspond to complex numbers, however, the positive eigenvalues represent the Euclidean components of the neighborhood relationships. Empirical results show that the positive eigenvalues alone give a good representation of the spatial relationships (Borcard and Legendre 2002, Borcard et al. 2004). I used both Euclidean distance (PCNM-E) and stream channel distance (PCNM-S), to

construct two sets of spatial eigenvectors using the basic PCNM methods (Table 3), to test the importance of overland, direct distances and in-stream distances on structuring water quality and in-stream conditions, considering all sites as connected (Table 4 and see Table 3 for distance matrix construction methods).

This basic framework has been developed to include options for directional and weighted spatial representations with the potential to accurately represent processes in stream networks (Blanchet et al. 2008a, Dray et al. 2006). Another set of spatial eigenvectors were created using the Moran's eigenvector map (MEM) method (Table 3). This involved the diagonalisation of a spatial weighting matrix, constructed through the Hadamard product between two previously computed resemblance matrices, a binary connectivity matrix and a weighting matrix (stream distance between sites) (Dray et al. 2006). This allowed variable connectivity and ease of travel between sites to be taken into account in the construction of this metric (MEM, Table 4). Finally I used average discharge to weight the links between sites to represent the rate at which solutes and particles in the water are transferred throughout the system. The weighted links table was then multiplied by the matrix of stream distances (as in MEM with the connectance matrix) to create corresponding weighted metrics (MEM-W Table 3). Eigenvectors (constructed by any method) with large eigenvalues describe global structures whereas those with small eigenvalues describe local structures (Borcard and Legendre 2002).

The spatial metric that best described variation in water quality and in-stream conditions was chosen using the adjusted coefficient of multiple determination (R^2_a) to compare the variance explained by each (Peres-Neto et al. 2006, Blanchet et al. 2008a). Each set of eigenvectors from each of the eigenfunction analyses for each distance measure, was subjected to forward selection ($\alpha < 0.1$) to detect eigenvectors explaining the most variance in water quality (Blanchet et al. 2008a). The multivariate set of physico-chemical water quality variables was then analysed as functions of the set spatial eigenvectors by canonical redundancy analysis (RDA), a multivariate regression-

based analysis using the spatial vectors as predictors of water quality (Dray et al. 2006, Peres-Neto and Legendre 2010). The spatial metrics were then assessed as to their ability to describe spatial structures within each system by comparing the R^2_a. The level of spatial structuring was also compared between the five systems using the R^2_a associated with the best spatial descriptive metric (Blanchet et al. 2008a).

These analyses were conducted in the R-language environment (R Development Core Team 2007) using the packages "vegan" (Oksanen et al. 2007) for RDA and variation partitioning, "PCMN" (Dray et al. 2006) for the construction of PCNM variables and "packfor" (Dray 2005) for the selection of explanatory variables in the RDA. In all tests of significance, 999 permutations were used. Following Anderson and Legendre (1999) permutation of raw data were adequate for ANOVA as there are no outlier values in the factors.

Table 3. Four different spatial metrics were tested on sites in each of the five systems. Eigenfunction-based spatial filtering techniques were used, allowing flexibility in weighting and directionality of spatial representation. I used PCNM with two distance metrics, Euclidean distance and stream distance. A third spatial metric based on Moran's eigenvector maps (MEM) used stream distance between flow-connected sites to account for the dendritic nature of site-wise connectivity. A corresponding weighted metric was constructed based on the average velocity encountered between sites.

Distance metric	**Weight**	**Code**	**Description**
Euclidean distance		PCNM-E	PCNM on Euclidean distances between sites
Stream distance		PCNM-S	PCNM on stream distances between sites
Stream distance between flow connected sites only		MEM	MEM all sites directly connected by flow in both directions
	Average discharge	MEM-W	MEM – weighted by average discharge encountered between sites.

2.2.5.4 Potential drivers of system level spatial autocorrelation

The R^2_a associated with the best spatial descriptive metric was considered to represent the level of spatial autocorrelation within each system. To investigate the causes of varying levels of spatial

structure in stream water quality, the best R^2_a each network was compared to a series of system level parameters (Table 2) using simple linear regression in the R-language environment (R Development Core Team 2007).

2.3 RESULTS

2.3.1 Variance in physico-chemical variables associated with stream size

No strong trends were evident in median water quality value between the four stream orders examined, when compared across all sites and systems. Due to the non-independence and lack of normality of the data p values were not tested statistically. It is clear, however, that there was more variation about the median in the first and second order sites than the third and fourth, across all variables (Figure 2a-f).

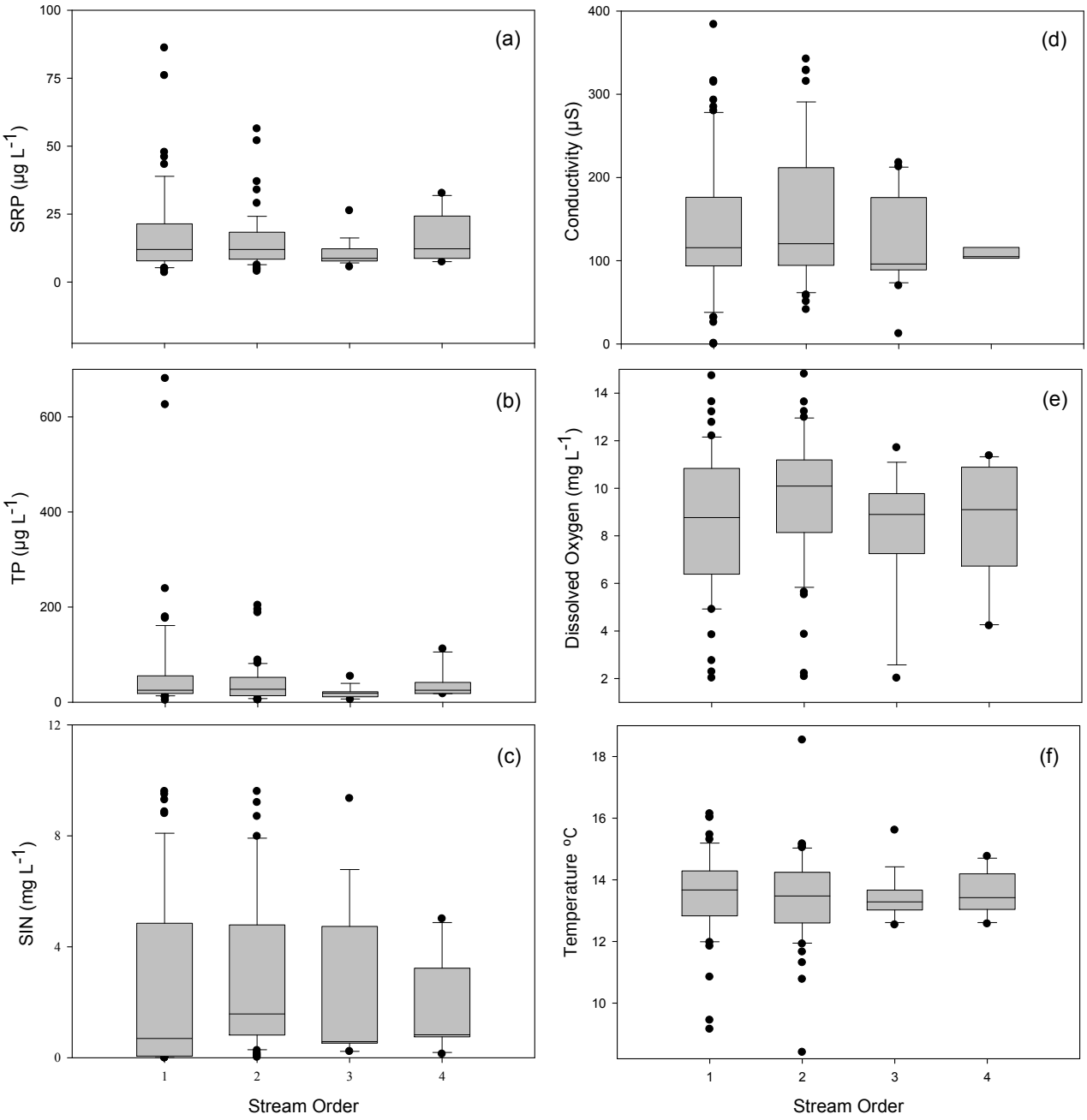


Figure 2**.** Variation in soluble reactive phosphorus (a), total phosphorus (b), nitrate (c), conductivity (e), dissolved oxygen (e) and temperature (f) across all sites and systems by stream order. The shaded boxes show the spread of data to the 25th and 75th percentiles around the median value, lines extend out to the 5th and 95th percentiles and dots are outliers.

Quantile regression was used to evaluate the variance structure across the range of the data set. Variability in physico-chemical PCA axis one and two scores declined with increasing stream width (Figure 3). Axis one described the variability in dissolved oxygen, pH, temperature and

conductivity. The slope of the 10th quantile regression line is significantly different from zero, indicating a positive floor, limit relationship (p = 0.005, Figure 3a). The variation in the upper level of axis two scores, which represented concentration of dissolved oxygen, SRP, TP and turbidity, significantly reduced with increasing stream size (p =. 0.01755, Figure 3b). Water quality in small streams was highly variable when compared with the homogeneous nature of larger streams.

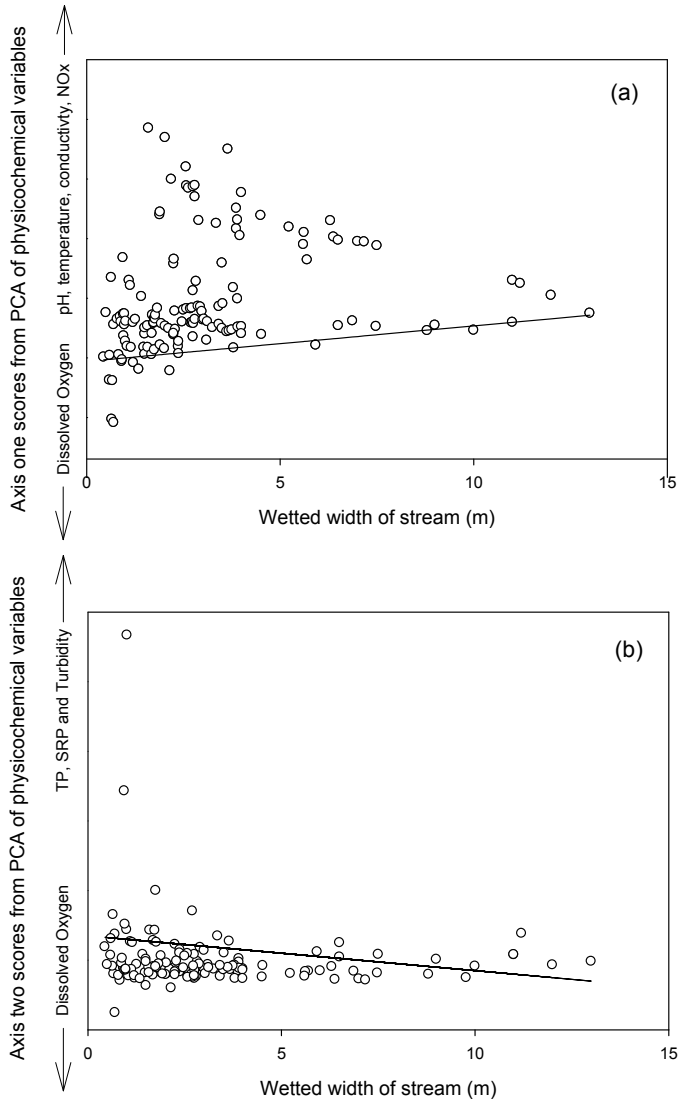


Figure 3. Scatter plot and quantile regression fits of the principal components analysis axis one and two scores, from the Standardised PCA of the physico-chemical variables against stream wetted width. Variables significantly correlated with each axis are displayed. Superimposed are the 10^{th} (a) and 90^{th} (b) quantile regression lines.

2.3.2 Spatial patterns in system-wide water quality

The panels below show the high level of variability in water quality that exists within a single system, and between two systems (Figure 4). Taking Birdlings Brook and the Cam River as examples, one can see that each system demonstrates patterns that are unique, with respect to each physico-chemical variable, and that these patterns are not always consistent between these two systems (Figure 4). Dissolved oxygen shows lower levels in the tributaries than main stems of Birdlings Brook, whereas branch specific dissolved oxygen levels occur in the Cam River (Figure 4a). Total phosphorus decreased down the main stems of Birdlings Brook, yet, comparably, is low throughout the Cam River. A tributary high in total phosphorus enters the Coldstream branch of the Cam River, yet this branch remains low in total phosphorus below this confluence (Figure 4b). DIN had a patchy distribution in Birdlings Brook, while again in the Cam River the distribution was branch specific (Figure 4c)

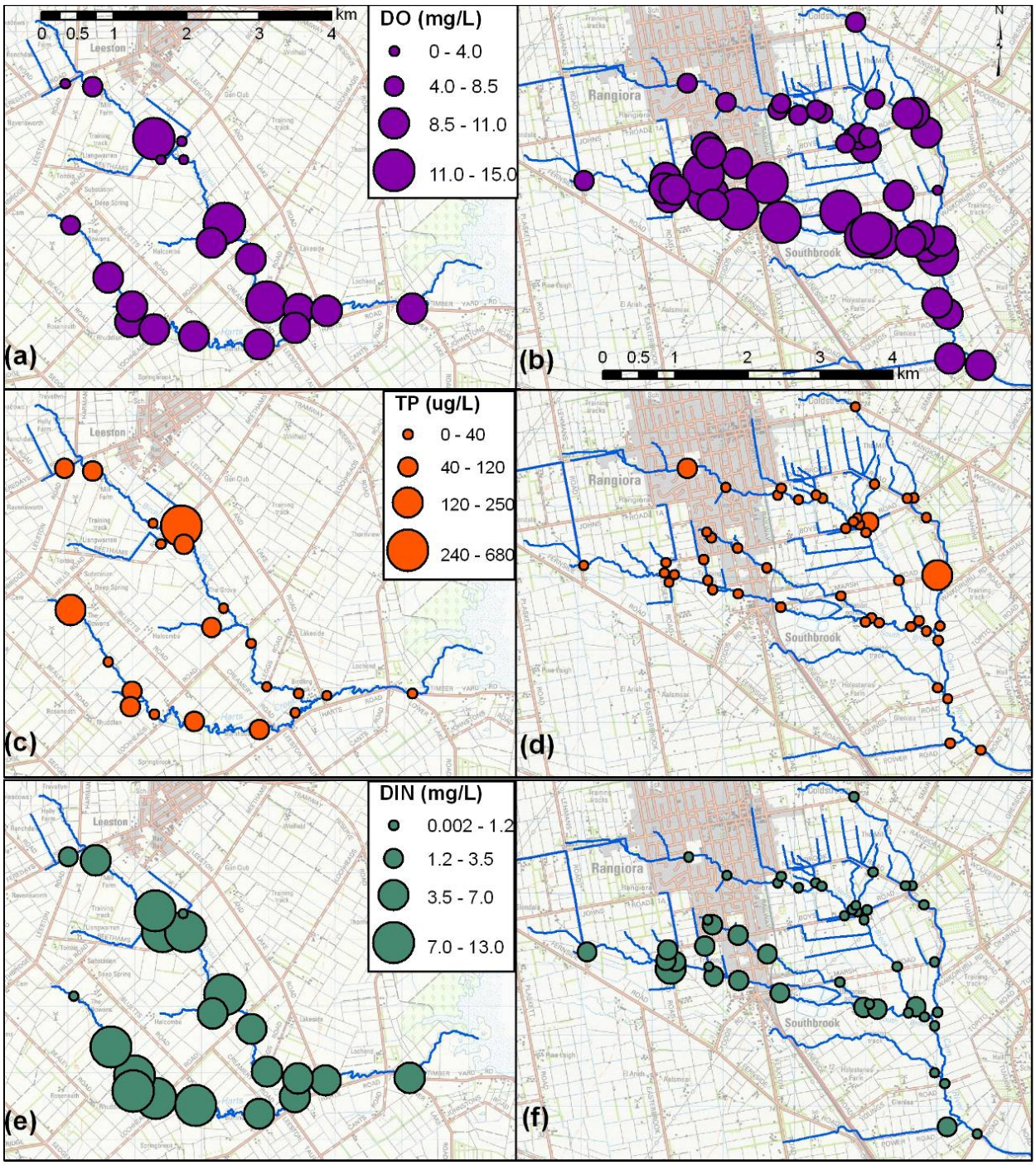


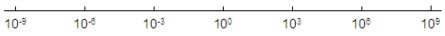
Figure 4. Graduated symbols represent spatial variation in dissolved oxygen (a and b), total phosphourous (c and d) and dissolved nitrate (e and f) at each site in Birdlings Brook (left panels) and the Cam River (right panels). The same key is relevant across both systems for each variable. Sites are overlain on the mapped streamlines and topographic map images. The catagories of each variable were defined using the "Jenks" natural break method (ArcMap 9.2).

2.3.3 Comparing metrics and levels of spatial autocorrelation

The spatial variables created by MEM method described the most spatial variation in three of the five systems, while those created by the PCNM method using stream distance performed the best in the remaining two systems (Table 5, Figure 5). Either MEM or MEM-W performed relatively well in all systems (Table 5, Figure 5). However the order of performance of the spatial metrics differed between the systems. Euclidean distance generally explained relatively little spatial variation, however in Birdlings Brook and the L2 it did not perform quite as poorly as in the other systems (by comparison within each) (Table 5, Figure 5). Birdlings Brook had the highest level of spatial structuring of all five systems (the largest R^2_a values across all metrics), followed by the Cam River. This means that sites that are close together have similar levels of water quality, compared with sites that are far apart. The Styx River had the least amount of spatial structuring in water quality, with no significant spatial vectors being chosen at all from the PCNM-E method (Table 5). Hunters Stream differs from the other systems in that the metrics are not evenly spread in their ability to describe the spatial pattern. Despite having the least spatial pattern in general, described by three of the metrics, Hunters Stream displays a moderate amount of spatial structure compared to the other systems, when described by the MEM weighted metric (Table 5, Figure 5).

In general the significant spatial PCNM vectors are fewer and describe larger spatial structures than the MEM vectors, which cover a broader spatial range (Table 5). The L2 had spatial structuring at the finest scale, as well as at a large scale, while Hunters Stream only at the large scale and the Styx only at an intermediate scale (Table 5). Birdlings Brook and the Cam River had significant spatial vectors that described a wide spread of spatial scales (Table 5).

Table 5. Differing amounts of variance in water quality composition were explained by the four spatial metrics in the five stream networks. This was determined using redundancy analysis (RDA) with forward selection (α <0.1 and 9999 permutations). The '# sig. vectors' indicates the number of vectors that were significant during forward selection, while 'variance' is how much variation in community composition this set of variables explained. The most significant distance metric for each stream is highlighted in bold with probability (P) values indicated. R^2_a is the correlation coefficient, adjusted for the number of variables in each model so that they can be compared robustly. Refer to Table 2 for distance measurement acronyms.

	# sig. vectors	Variance	P	R^2_a	Ranges of sig. spatial vectors (eigenvalues on Log scale)
L2					
PCNM-E	2	0.20	0.015	0.13	
PCNM-S	**3**	**0.34**	**0.005**	**0.25**	
MEM	7	0.42	0.015	0.20	
MEM-W	3	0.17	0.11	0.06	
Birdlings Brook					
PCNM-E	5	0.57	0.015	0.43	
PCNM-S	2	0.41	0.005	0.34	
MEM	9	0.82	0.005	0.67	
MEM-W	**8**	**0.88**	**0.005**	**0.81**	
Hunters Stream					
PCNM-E	1	0.08	0.16	0.03	
PCNM-S	2	0.17	0.053	0.09	
MEM	2	0.11	0.17	0.04	
MEM-W	**10**	**0.65**	**0.005**	**0.39**	
Cam River					
PCNM-E	5	0.30	0.005	0.22	
PCNM-S	7	0.36	0.005	0.26	
MEM	21	0.71	0.005	0.46	
MEM-W	**26**	**0.76**	**0.005**	**0.47**	
Styx River					
PCNM-E	0	NA	NA	NA	
PCNM-S	**3**	**0.31**	0.026	**0.19**	
MEM	2	0.26	0.01	0.18	
MEM-W	2	0.25	0.015	0.17	

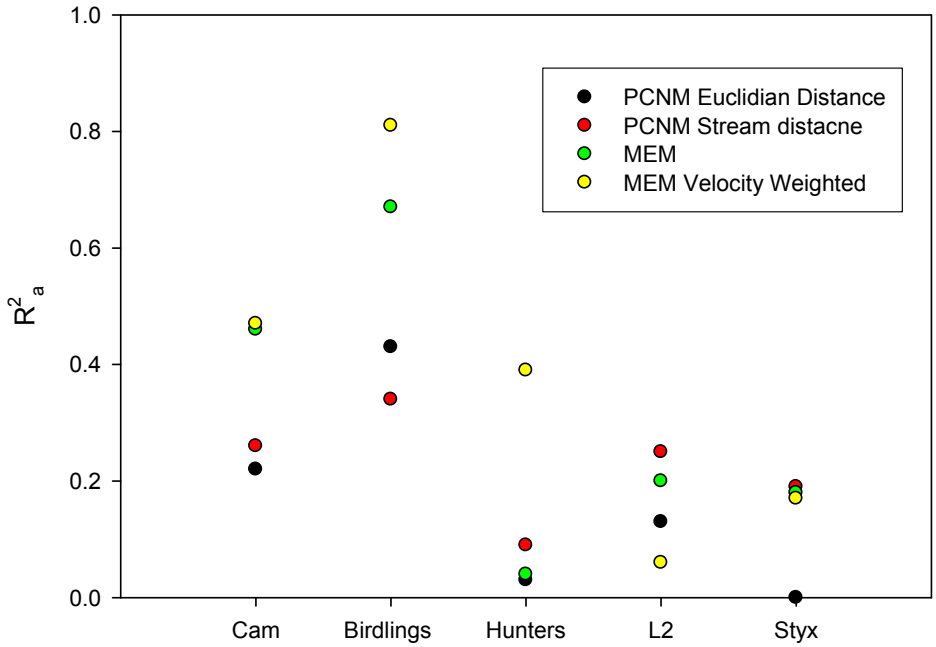


Figure 5. Differing amounts of variance in water quality were explained by the four spatial metrics in the five stream networks. This was determined using redundancy analysis (RDA) with forward selection (α <0.1 and 9999 permutations). R^2_a is the correlation coefficient, adjusted for the number of variables in each model so that they can be compared robustly. Refer to Table 2 for distance measurement acronyms and text for detailed methods.

2.3.4 Testing for spatial autocorreletaion in the systems

A broad range of system level properties were tested against the level of spatial autocorrelation present in each system, described by the R^2_a value associated with spatial metric that described the most variance in water quality. The amount of variance in water quality explained by the best spatial model in each system significantly decreased as catchment area increased ($R^2 = 0.81$, $p = 0.037$, Figure 6d). The proportion of forest cover in the 100m-buffer zone and proportion of the catchment area that was confined aquifer also had negative relationship with of variance in water quality explained spatially, however these were weaker relationships than that of catchment area, significant at α=0.1 ($R^2 = 0.56$, $p = 0.087$ and $R^2 = 0.3749$, $p = 0.097$ respectively, Figure 6a and b). The number of springs in each catchment area, and both measures of dendricity, (drainage and

confluence density) had no significant influence on the level of spatial autocorrelation in the five systems I studied (Figure 6c, e and f).

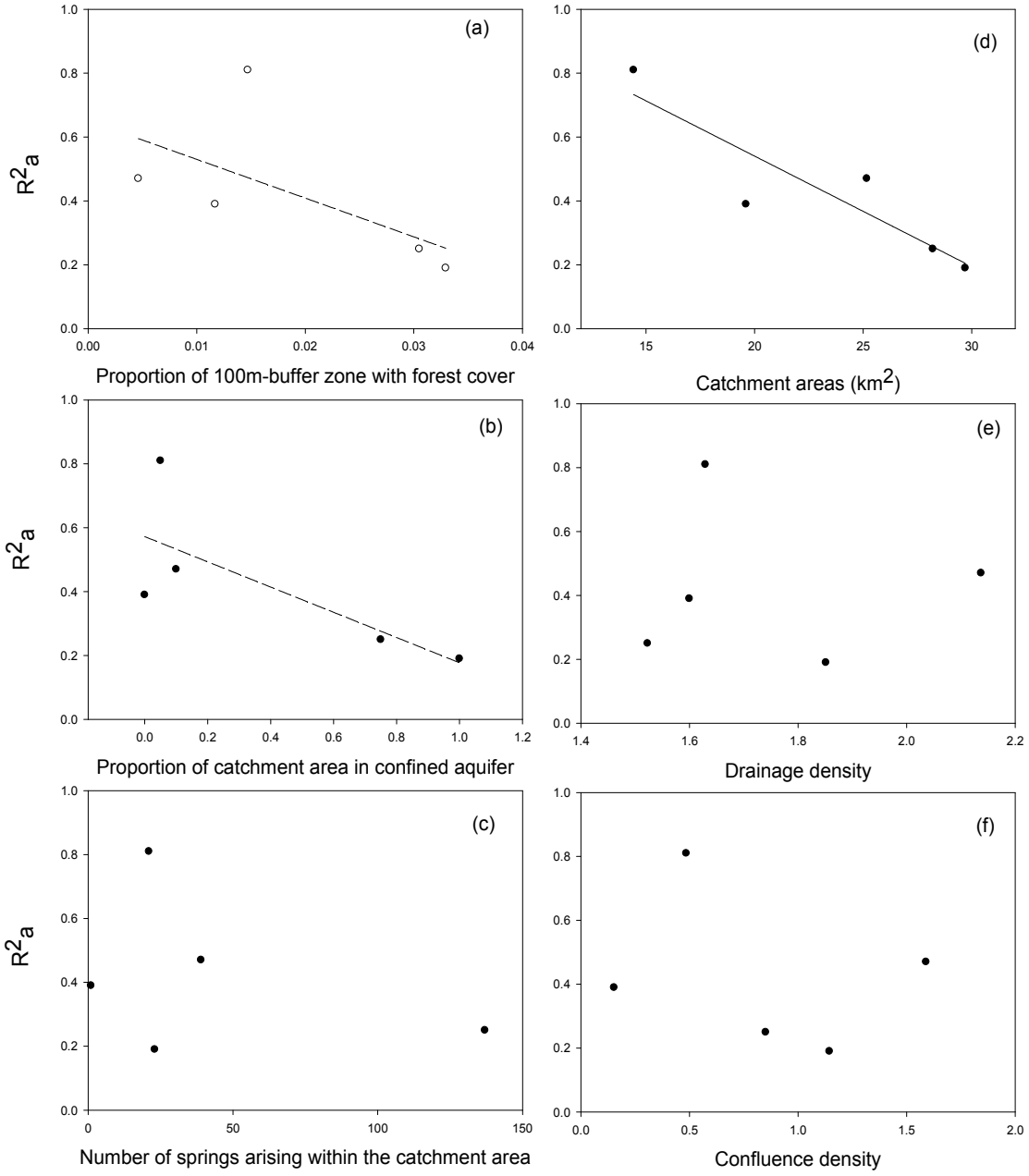


Figure 6. The amounts of variance in water quality explained by the best spatial metrics (using the associated R^2_a) in each of the five stream networks compared with system level properties, proportion of the 100-m buffer zone with forest cover (a), proportion of catchment area in confined aquifer (b), number of springs within each catchment (c), catchment area (d) and confluence and drainage densities (e and f) (see table 2 for derivation of these). Regression lines displayed where the general liner model indicted a significant relationship (the solid line is significant at α=0.05 and dashed lines at α=0.1)

2.4 DISCUSSION

Water quality is an important metric of stream health; however, variability of surrounding land-use and innate features of the stream network alter the pattern of water quality in a system. I found that headwater streams were most affected by surrounding conditions and had the greatest variability in water quality. Network characteristics in the five stream systems led to different levels in spatial structure between streams. Specifically, spatial variation in nutrient inputs (via groundwater inputs and runoff) and in-stream nutrient processing may alter the degree of spatial autocorrelation within stream networks. The degree of spatial structure will affect how headwater effects on water quality are proliferated through the stream network.

2.4.1 Pervasive variance in the water quality of small upstream sites

Using several metrics of stream size, I found that small stream sites had more variable water quality than larger stream sites. Thus, by extension, sites on tributaries, further up the network were more variable than main stem sites, nearer the outflow of the network. A similar trend was found for five out of the eight water quality variables in low elevation, pastoral streams, from throughout New Zealand (Larned et al. 2004) and in long term monitoring of the Seine River in France, where the range in conductivity values decreased with increasing stream order (Meybeck et al. 1999). The change in variance structure with stream size may help explain why limnological data rarely conform to parametric assumptions (Matthews et al. 1991). For this reason, systematic trends and causal relationships with stream size have often been difficult to detect, even when present in water quality data (McKee et al. 2001, Larned et al. 2004). Heterogeneous errors seem to be pervasive in water quality data, resulting in the need for transformation and high levels of unexplained variation (Jones et al. 2001, McKee et al. 2001, Larned et al. 2004). The identification of gradients along

streams, by order, has been successful for biotic parameters, but often not for water quality (Beecher et al. 1988, Naiman et al. 1997). The pervasiveness of this variance is demonstrated in heterogeneity in water quality that occurs within single catchments, in pristine condition (Clow et al. 1996, Eyre and Pepperell 1999, Finlay et al. 2011).

Water quality in headwater streams is heavily influenced by the nature of the surrounding landscape, the characteristics of local and regional groundwater inputs, degree of riparian buffering, and in-stream processes. Due to their small size, headwater streams are particularly sensitive to variations in these controlling factors. The nature of stream networks in the Canterbury lowlands (i.e. dendritic network morphology) is such that headwater streams are widely distributed throughout the landscape and are exposed to a wide range of land-use, hydrologic and biological conditions. This higher sensitivity and wider exposure lead to the considerable variation in water quality that I observed in headwater streams.

Small streams are more strongly influenced by local land-use conditions than larger ones (Buck et al. 2004), this is likely to enhance their variability (Bishop et al. 2008). This can be explained by the higher edge to volume ratio compared with large streams (Bishop et al. 2008). This is one of the principles behind sampling water quality at base-flow, when the mean transit time of water (thus, contact with the landscape) is highest at low flow; so it is thought that water quality will be tightly coupled with land use (Biggs et al. 1990, Lyon et al. 2008). In small streams, the smaller volume of water is less able to dilute the diffuse influences of surrounding land and any point source influences, compared with larger discharge sites. Whereas, sites further down a system, with a higher flow volume, are more able to buffer or dilute the impacts of incoming polluted tributaries (for example, the tributary high in total phosphorus (discussed in the results) has no impact at downstream mainstem sites).

A further explanation for the small stream variability I found on the Canterbury Plains is the general heterogeneity of landscape conditions through which each network passes. Despite the flat

appearance of the Canterbury Plains, with increasing urban sprawl, the dividing up of large farms into smaller mixed-use land and lifestyle blocks, as well as pockets of vegetation restoration, the Plains are far from homogenous. Soil pattern on the plains can be deduced from geomorphic history and are more complex than previously thought. Remnants of old alluvial fans are sometimes preserved in association with small rivers (Webb 2008) and many soil types exist within each of the five catchments studied (Environment Canterbury GIS layer). This could lead water to come in contact with sediment and soil of variable characteristics and redox conditions can alter biogeochemical pathways and the patterns of stream water quality (Dahm et al. 1998). Additionally, different types of land-uses can have unique influences on individual water quality parameters (Jones et al. 2001, Jones et al. 2004) and extremes of each variable are more likely to exist in first-order streams. In small streams, the contact time between nutrients and the water column is also high, due to the low flow rate. Thus, as well as being perceptive of the local land use, small streams can be particularly effective in retaining nutrients due to shorter uptake length for nutrient spiralling compared with large streams (Peterson et al. 2001, Niyogi et al. 2004). Thus, their variability in biological conditions may lead to spatial and temporal variability in water quality of lower stream orders.

Large scale variation is also likely to exist in the interaction of surface and ground waters on the Plains. Confined and unconfined aquifer matrices exist in a spatially variable, alternating form that has been described as “beads on a string” (Cruezé des Châtelliers et al. 1994). Many microclimates exist on the Canterbury Plains that could potentially add to and exacerbate the effects of groundwater variability (Sturman 2008). Given that a significant amount of variability has been found in the water chemistry of a single catchment, with no bedrock variation and or human impact (Clow et al. 1996), it is reasonable to expect much variability on the Canterbury Plains.

Adding to this effect, riparian management is inconsistently practised across the Canterbury Plains (Greenwood, McIntosh and Harding, unpublished data). Small streams are often unfenced to

allow for stock access to drinking water and have had any remaining riparian buffer removed to increase productive land. Stream banks often collapse adding sediment and enhancing transport of pollutants directly into the waterways (Tabacchi et al., 1998, Parkyn et al., 2003, Baker et al. 2007). Several studies have found that surface runoff only enters a river system through a relatively small proportion of the total riparian length (McGlynn and Seibert, 2003) and have observed decreases in surface runoff with increasing stream order. This, along with their unbalanced contribution to system length, has lead several authors to conclude that riparian zones along headwater streams may be disproportionately important for nutrient retention (Dosskey et al. 2005).

2.4.2 Identifying system level patterns using spatial autocorrelation – untangling the variance

A tendency towards positive spatial autocorrelation at small scales makes intuitive sense. Sites that are close together on the stream are similar. Sites begin to show no spatial autocorrelation (randomness) or negative spatial autocorrelation (homogeneity) once they are a certain distance apart (Urban, 2003). Measures of spatial autocorrelation are a useful way of summarising variable spatial patterns. Some level of spatial autocorrelation is present in most stream water quality data due to the directional nature of flowing water (Lyon et al. 2008, Tu and Xia 2008). The presence of a connection by flow and the distance along the stream channel between sites potentially has a greater influence on spatial patterns in water quality than the distance directly, or "as the crow flies", between sites (Fagan 2002). This was the case in all five systems I studied, where distance metrics based on stream channel distance consistently explained more variance in the spatial pattern than the direct (Euclidean) distance. Therefore, I conclude, as was found in several recent studies, that stream channel distances are more suitable for use in water quality studies than Euclidean distances. Euclidean distance fails to accurately represent the spatial configuration, connectivity and relative position of sites in a stream network (Peterson et al. 2007, Lyon et al. 2008).

Increasing heterogeneity of water quality with increasing separation between sites along a stream can arise from changes in the local land use and point source inputs, such as tributaries (but also unidentified water entering the system - piped outfalls or ground water upwelling). Changes in riparian and in-stream conditions (macrophyte or algal cover) may in turn lead to differing levels of nutrient spiralling. Any disruption to the balance between these processes, leading to net retention or regeneration of nutrients, may disrupt the level of pattern spatial similarity within a stream system. Thus, the extent of spatial similarity within a system can be used to make inferences regarding the occurrence of these processes.

One concept worth applying is that buffer zones can be either leaky or retentive, changing the connectivity between the surrounding landscape and the stream (Baker et al. 2007). Vegetated buffer zones can act as a sink for nutrients (Billen and Garnier 2000), insulating the stream channel from the impacts of surrounding land use. Water quality would be expected to be more spatially similar under such "closed" conditions, relying on in-stream processes to regulate water quality. Reaches without intact riparian buffers are more "open" to impacts from the surrounding land, relying more on terrestrial connectivity than aquatic biology. Such factors may alter water chemistry at similar sites differently than would be expected for their degree of physical separation and chance alone.

The high level of spatial structuring of the water quality within Birdlings Brook and the Cam River could be due to the fact that these streams are well insulated from the surrounding landscape and/or demonstrate balanced in-stream nutrient processes. The widely spread eigenvalues of the spatial vectors, which best modelled the water quality variation in these systems, indicated that processes may be balanced at multiple scales. However, my results indicated a negative relationship between the amount of spatial autocorrelation and the proportion of the buffer zone that is forested. At the scale of the Canterbury Plains, only coarse scale forest cover data is available, this may not

be an accurate indicator of the actual riparian buffer conditions in these systems, particularly for smaller streams.

The L2 and Styx Rivers have comparatively less spatial similarity in water quality than the Cam River and Birdlings Brook. The L2 and Styx River systems flow through damp wetland areas and, as they are situated on the edge of confined and unconfined aquifers, upwelling water produces many springs that contribute to their flow. The spatial patterns may be less evident in these systems, as they are interrupted by an influx of groundwater of different quality from the surface stream water. Streams on the Canterbury Plains are likely to vary in their degree of openness with regard to connectivity with groundwater. Across the five systems that I studied, the proportion of each catchment that was on a confined aquifer explained the observed patterns, but the number of springs did not.

Hunters Stream is unusual in that it exhibits very little significant spatial correlation, other than when I examined it using the discharge weighted measure. The channels of this once-natural stream are now fed almost entirely by water from stock water races. However, discharge decreases longitudinally, which provides an explanation for the discharge weighted spatial metric being the only metric that explained any significant level of the variation in this steam. The low flow levels in Hunters Stream may mean that spatial patterns are only evident in those parts of the stream where flow is higher. In these parts, the flow can transport nutrients or disturbance effects before the water is lost from the channel. As the earlier results of this chapter indicate, water conditions are likely to be highly variable in small streams, thus a variable pattern throughout the system may also result from its nature as a small stream, based on the potential mechanisms discussed earlier.

One stream network characteristic that should have regulated spatial structure was branching pattern. Specifically, I expected higher levels of dendricity would lead to decreases in the spatial similarity of water quality. This decrease would potentially be caused by the confluence of many tributaries, of differing water quality, disrupting longitudinal similarities. This effect would be

exacerbated by the nature of dendritic systems, with their capillary of small streams, branching throughout a landscape of potentially great variability (Bishop et al. 2008). Yet, in the five systems I studied, there was no relationship between spatial autocorrelation in water quality and dendricity. The exact cause of spatial structuring in water quality in each system considered in this study could not be absolutely defined, as my analyses did not distinguish between the spatial dependence of water quality (i.e. spatially structured environment) and the actual spatial processes driving community structure (e.g. the downstream flow of solutes). However, my results confirm the usefulness of the 'space as a surrogate' approach (McIntire and Fajardo 2009) for identifying areas where landscape interaction or changes in in-stream conditions potentially occur. The interaction of spatial processes with catchment land-use, riparian and in-stream processes, in influencing water quality, was addressed using a focus on the Cam River in Chapters Two and Three.

2.4.3 The role of small streams in spatially correlated systems

In stream systems where water quality is spatially correlated, any disturbances at the headwaters would proliferate through the system to a greater degree than in a system with less spatially correlated conditions. My results indicated that the water quality of headwater streams can have a disproportionate influence over water quality throughout an entire network. Care should be taken during the intensification of land bordering small streams, as the impact of a disturbance may reach far beyond the area of land under development. Whereas, the protection of headwater streams may provide a cost-effective focus for management of catchment water quality, rather than protecting entire systems. Management of small streams is particularly important, not only due to their spatial control, but also due to their highly variable water quality within the lowland areas of the Canterbury Plains.

CHAPTER THREE - CONSISTENT SPATIO-TEMPORAL WATER QUALITY VARIATION IN A COMPLEX RIVER SYSTEM

The Northbrook: a branch of the Cam River, which flows through dairy farms to the East of Rangiora.

3.1 INTRODUCTION

The physical and chemical composition of river water strongly influences its suitability for both aquatic life and use by humans (Davies-Colley and Willcock 2004). Early interest in water quality arose from a geological perspective, in relation to catchment geolithology (Walling and Webb 1975). However, with increasing degradation of freshwaters by agriculture and urbanisation, both worldwide and in New Zealand, it is important to understand variations in fresh water quality in terms of multiple landscape variables (Jones et al. 2001, Allan 2004). Water quality is a valuable indicator of the degree to which a river system has been impacted by changes in the environment (Hem 1985, Jones et al. 2001). Yet several aspects of limnological data sets make them notoriously difficult to deal with in analyses (Walling and Webb 1975, Matthews et al. 1991).

3.1.1 Difficulties in the analysis of water quality data

Limnological data are generally non-linear, rarely conform to parametric assumptions and are often measured using incommensurable units such as length, concentration, and frequency (Matthews et al. 1991). In addition, most water quality research generates incomplete data sets through sample loss and sampling design (Matthews et al. 1991). Gradients and causal relationships may be difficult to detect due to the often high variability in water quality data (McKee et al. 2001, Larned et al. 2004). Heterogeneity in water quality even occurs within single catchments in pristine condition (Clow et al. 1996, Finlay et al. 2011). Stream water quality also varies temporally, largely through changes in flow rate (Salvia et al. 1999, Pionke et al. 1999, Wayland et al. 2003).

Most, if not all, environmental data are also spatially correlated (Legendre and Fortin 1989). However, this is a particularly significant problem in rivers, where the directional nature of flowing water means that data collected are often spatially correlated and non-independent (Lyon et al. 2008, Tu and Xia 2008). Two main problems have been identified. Firstly, spatial autocorrelation; where

one site may have more similar water quality to a site nearby, than to another site far away (Legendre 1993, Fortin and Dale 2005). This is because sites near each other may be affected by the same human influences, climate, geology and may even be connected by flow. Secondly, spatial non-stationarity; where the relationship between the independent and dependent variables are not constant over space and time (Fotheringham et al. 2002). This often occurs in water quality data as local conditions are likely to vary (Tu and Xia 2008). Relationships that exist in part of the study area may be obscured by an apparent overall trend or lack thereof. These situations result in spatial autocorrelation and heteroscedasticity in the residuals of analyses, which reduce their power to detect causes of water quality variation (Legendre 2002).

Under positive spatial autocorrelation, the value of a parameter, at a point, can be predicted in part from values at surrounding points. As values are not stochastically independent, in reality each point brings less than 1 degree of freedom to the analysis, reducing its power (Legendre 1993, Fortin and Dale 2005, Rangel et al. 2006). Thus, the assumptions of normality and homoscedasicity are often ignored in water quality studies and many authors rely on the robustness of statistical tests to identify significant trends, despite violation of fundamental assumptions (Matthews et al. 1991, Tu and Xia 2008, Rangel et al. 2010).

Within stream networks, which are inherently complex, it is particularly difficult to detect water quality relationships due to stochastic variability, natural and human induced spatial heterogeneity and logistical challenges of working across multiple sites (Prowse 1984, Finlay et al. 2011). The high level of correlation between land use variables within a catchment is problematic for their use in regression analysis (Jones et al. 2001). Water quality is usually monitored on a regular basis at a small number of sites in a watershed, usually at the outlet of a larger basin, thus all non-point and point sources are integrated and mixing of waters may obscure local extremes (Walling and Web 1975, Salvia et al. 1999). Yet effective management involves identification of the key sources and processes. Untangling the effects of geology, topography and land-use on spatial

variation in water quality is also difficult, as these variables are often correlated, so one may obscure the impact of another as in community ecology (Walling and Web 1975, Prowse 1984, Laliberté et al. 2008). Temporal variations in water quality may also contain temporal autocorrelation, which can further convolute studies which are conducted across space, but often through logistic constraints, not conducted approximately simultaneously (Fortin and Dale 2005).

3.1.2 Investigating variability using a spatial-temporal snapshot

Stream ecologists have long studied biogeochemistry under a whole catchment perspective (Webster et al. 1979, Deitrich et al. 1982). Recent work on the dendritic and hierarchical nature of streams has highlighted the need to pay more attention to spatial structure and connectivity (France and Duffy 2006, Ganio et al. 2005, Brown and Swan 2010). Moreover, many insights can be gained by spatial studies of stream physicochemistry, especially if they incorporate temporal components. Identification of the flow-on effects of changes across a wide range of scales, and predicting their consequences, is an important topic for ecological research in general (Ludwig 2007). Not only did early work on catchment level water quality variation fail to take these issues into account, elements of spatial autocorrelation were overlooked in modelling and analysis (Brown and Swan 2010). Although researchers of physico-chemical processes pioneered the inclusion of spatial perspective (Hubbard Brook work and snapshot studies in the 1990's), they have not benefited from the more recent development of statistical methods capable of handling the problems associated with spatial issues.

In all ecological fields, the interest in variability associated with landscape and space has developed faster than the appropriate statistical methods (Rangel et al. 2006). A literature review found that 80% of studies which analysed spatial data did not use space explicitly in their analysis (Dormann 2007). Many researchers continue to use traditional statistical methods, despite knowledge of the spatial nature of their data and new methodology developments (Herlihy et al.

1998, Billen and Garnier, 2000, Jones et al. 2004, Finlay et al. 2011 and many more). Traditional methods are not well equipped to deal with spatial autocorrelation, so water quality scientists risk drawing incorrect conclusions (Legendre 1993, Dormann 2007).

Perhaps confused by the large number of statistical models and fitting methods available, many ecologists believe space is best left to the specialists (Beale et al. 2010). However, one such spatial method, principal coordinates of neighbour matrices (PCNM – Borcard and Legendre 2002), shows potential to unravel and quantify the processes that drive spatial and temporal variation across multiple scales, in many fields of biology (Brind'Amour et al. 2005, Lacey et al. 2007, Laliberté et al. 2008). The application of the PCNM approach to spatial dependence of in-stream flow variables demonstrates its potential use in a variety of water resources studies (Lacey et al. 2007). Furthermore, Thompson and Lake (2009) stress that, as streams are unique in their openness, linearity and directionality; the movement of energy, nutrients and matter are important. Thus, more research is needed on spatial patterns of nutrient retention and processing in streams (Thompson and Lake 2009).

The Cam River system is suitable for the study of spatial and temporal variation for a number of reasons. Worldwide, the study of within-catchment variations of water quality has tended to take place in relatively pristine catchments with little variation in land-use (Clow et al. 1996, Finlay et al. 2011). The study of degraded catchments such as the Cam River presents difficulties in the untangling of multiple impacts at complex scales. Essentially, spatial and temporal heterogeneity is the basis of most ecological studies into relationships; increased heterogeneity provides the opportunity for discovering a greater variety of effects (Legendre 1993, Fortin and Dale 2005). Complex systems are becoming increasingly common with anthropogenic impacts. Such systems are often at risk for a variety of reasons, thus they deserve attention; perhaps to a greater extent than pristine systems. Possibly due to the impact humans have had on lowland areas in many countries, catchments of low relief are also under studied in this manner.

Lowland streams on the Canterbury Plains exist in a landscape which, with urban sprawl and intensification of farming practices, is becoming an increasingly complex mosaic (Winterbourn 2008). Lowland streams are also complex in terms of dendricity. They meander due to the low elevation, and thus make up a large proportion of flowing waterways within Canterbury, much of which exists as first order stems. Bishop et al. (2008) described this large capillary network of small streams as "aqua incognita", since the sheer magnitude of the length is a problem for gauging their variation. This lack of knowledge may come from the fact that we know much about a few headwaters (such as Hubbard Brook), yet processes in these systems cannot be applied to all streams (Bishop et al. 2008).

The study of first order streams needs attention due to the high contact time of water with the landscape. Small streams have subsequently been proposed as instrumental in conditioning the water for export, dampening flood waters, cycling nutrients (Bernhardt et al. 2003) or buffering pollutants (Klaminder et al. 2006), and for habitat and refugia (Lacey et al. 2007). Also, due to their high edge to volume ratio, processes in small streams are more sensitive to disturbance, including land use and climate change (Lowe and Likens 2005, Bishop et al. 2008). Therefore, knowledge of the small streams that make up the complex river systems on the Canterbury Plains could make a tangible difference to management of water quality in Canterbury and elsewhere.

This study characterised the spatial and temporal heterogeneity of water quality in the Cam River system under base flow conditions. I aimed to untangle the relative contributions of various spatial, landscape and in-stream factors to the observed patterns. This was done by conducting a series of snapshot sampling events. The impact of both spatial structure and environmental factors (catchment, buffer zone and in-stream) on water quality throughout the Cam River was assessed using the PCNM approach. The contribution of in-stream conditions and external inputs (diffuse or otherwise) was analysed specifically in relation to the impact of tributaries and changes in the longitudinal profile throughout the system.

3.2 METHODS

3.2.1 The Cam River system

The Cam River arises from numerous springs (39 found by Environment Canterbury to date, Environment Canterbury GIS layers) in and around the town of Rangiora, 25 km north of Christchurch in the South Island of New Zealand (43.3034°S 172.5914°E). These springs coalesce to form the four major branches; South Brook, Middle Brook, North Brook and Coldstream (Figure 1). The branches confluence in relatively quick succession to the South-East of Rangiora, forming the Cam River, which joins the Kaiapoi River and flows into the sea through the mouth of the Waimakariri River. The river is situated just off the edge of the large alluvial fan produced by the Waimakariri River (Webb 2008). Rangiora is a microclimate, due to the variety of wind interactions, that can be very hot in the summer and sometimes foggy and damp in other seasons (Sturman 2008). Riparian species naturally occurring in the area are flax, toetoe, grasses and sedges (Winterbourn 2008). The Cam River varies in its degree of naturalness throughout, from stoney-bottomed meandering channels to drains that are straightened and excavated. Water-races, fast flowing and of uniform depth, also enter this system. Parts of the river have abundant in periphyton and rooted macrophytes, which are periodically cleared by councils and farmers

The Cam River and its catchment have a long history of mixed use. The area was first settled by Europeans in 1851. At this time the river ran through native forest, wetlands and grasslands (Hawkins 1957). Several mills profited from the water in swamps and small creeks that drained into the Cam River. These produced a variety of wastes, many of which were disposed of into the river. Agriculture and farming became the primary industries after water from the swamps was drained and soils became more productive (Hawkins 1957). In more recent years, the non-urban parts of the

catchment have become dominated by dairy farming and lifestyle blocks (Biggs 1985, Agribase 2009).

All waterways flowing into the Cam River system were mapped; this included all drains and water-races or irrigation canals that entered the system. The actual flow path of the Cam River differs markedly from published delineations in several areas. Firstly the southernmost branch, as described by topographical maps and the River Environment Classification (REC, Snelder et al. 2005) does not flow into the Cam River, instead joins the Cust Main Drain. Secondly the water-race network intersects the system near the source of the Southbrook and Northbrook. Ground truthing revealed that this water-race network crossed the plains from the Waimakariri. An accurate GIS layer of all water flowing into the Cam River was built by tracing the river lines using Google Earth (Google Inc. 2009) images (the most accurate and high quality images available), in combination with field knowledge and ground truthing, using a GPS (Garmin GP560) (Figure 1). The complex dendricity and variety of land use within the Cam River catchment has the potential to reveal interesting spatial patterns. In particular, the long Northbrook branch provided an opportunity to study longitudinal effects on water quality.

3.2.2 Site selection

Base flows are most likely to be encountered in the late summer in New Zealand (Biggs et al. 1990). I sampled water quality in the Cam River synoptically, on a monthly basis, from January to March, in the austral summer of 2010. Sites were spread throughout the Cam River system as per the "snapshot" methodology to produce an instantaneous picture of all concentrations and fluxes in a watershed by sampling every confluence and discharge point within a short time period and where flow was stable (Walling and Webb 1975, Grayson et al. 1997, Salvia et al. 1999). Sites were placed where feasible on all first order tributaries and point sources entering the system (sources with flow less than 0.0005 m^3s^{-1} were not sampled). They were situated at least 25 m above and below every

point of confluence, so that no two sites were less than 50 m apart. Sites were also placed along uninterrupted main stem and tributary reaches, approximately 500 m apart (Figure 1). This method of site selection resulted in 81 sites throughout the network, of which a subset were sampled in the January and February sampling events, due to time constraints (Figure 1).

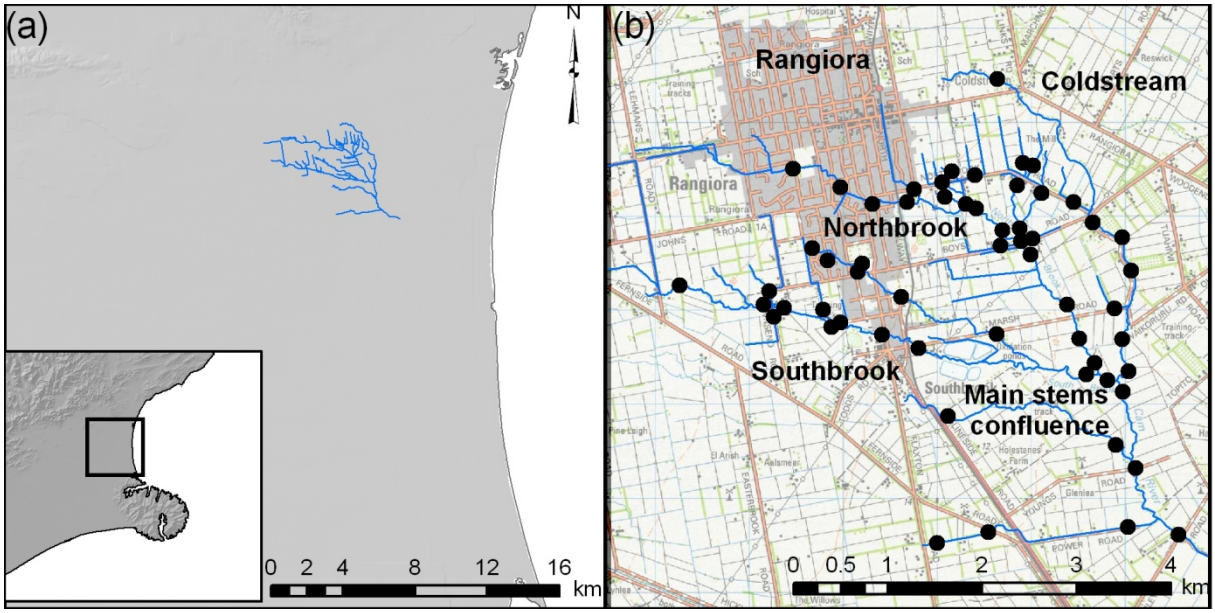


Figure 1: The location of the Cam River on the Canterbury Plains (a) and the location of the Cam River sampling sites (b). The blue lines show the natural channels of the Cam River system (constructed by tracing from Google Earth images and ground truthing). The stream drains towards the bottom right. The river lines and sampling sites are overlain on a topographical map of the area. The nearby town Rangiora and three major branches of the river are identified, these branches confluence in quick succession to the south of the town, forming the main Cam River

3.2.3 Snapshot Sampling

Samples were collected in three snapshot sampling events (Table 1), each spread over one to four days within a period of base flow, defined as the period between storms when the hydrograph was in the later stages of the recession limb, usually two days post storm flow peak (Pionke et al. 1999). Thus, sampling was not conducted if rain had occurred in the previous week in the catchment as well as in the Southern Alps or foothills which may cause the Waimakariri and Ashley rivers to rise,

raising the water table feeding the springs in the Cam River. Measured stage height variation at three sites indicates flow in Cam River was stable across each sampling period. Sampling was conducted up to 5 hours either side of midday to minimise diurnal variation in parameters such as dissolved oxygen and temperature. Diurnal variation is minimal for all major forms of nitrogen and phosphorus except for ammonia (not analysed here) (Finlay et al. 2011).

Sampling was more intensive than the survey conducted in the previous summer (Chapter Two) with 81 sites visited during the most intensive event. I measured eight physical and chemical properties of the water, as well as discharge, at each visit to a site. Other in-stream and site properties, which were considered more permanent, such as algal and macrophyte cover, were assessed on only one occasion. All variables, as well as field and laboratory methodology are described in Table 2. Each day I visited downstream sites first and collected water samples immediately on arrival at a site to avoid contamination from sediments disturbed by entering the water.

Table 1: Summary of the information relating to the three sample events in 2010 and the one-off sample event in 2009. Data are averaged across the days of sampling (NIWA, online climate data base, Rangiora station)

				Average across sampling run days (air temperature analysed for sampling hours only)							
	No. sites	Sampling period	Discharge at outflow	Day since rain >2 mm	Total rain	Soil moisture	Mean air temp	Max air temp	Min air temp	Sunshine amount	Evaporation (Penman ET)
Units		Day	m^3/s	Day	Mm	%	ºC	ºC	ºC	Hours	mm
January (2010)	63	2	1.44	5	0	14.5	17.8	21.2	15.6	5.4	4
February (2010)	71	1	1.59	6	0	13.1	20.5	26.8	10.3	7.7	3.4
March (2010)	81	4	1.18	5	0.1	13.3	16.8	21.9	4.7	2	2.1
March (2009)	47	6	1.21	3	0.2	15.3	13.5	26.9	7.1	6.9	1.6

Table 2: Field and laboratory methodology for the eight water quality variables and other predictor variables measured at each site, as well as treatment in analyses.

Variable	Units	Field and/or laboratory methodology	Reference	Treatment
Temperature	ºC	Time averaged spot measurement. The YSI Sonde 6600, a multi-parameter, water quality measurement device was placed in the thalweg of a run at the top of each site, continuously logging during time spent at each site. The final 5-20 minutes of data from each site was averaged (visit times varied depending on sample run intensity).		Eight response water quality variables
Dissolved Oxygen	mg/L			
pH				
Conductivity	µS			
Turbidity	NTU			
Dissolved Nitrite-N + Nitrate-N (DIN)	mg/L	I collected 80 mL of water in a syringe from the thalweg of the stream just below the surface and pushed through a filter (Whatman GFF 250 Millipore rating) into a 100mL opaque plastic bottle (pre-soaked in 5% hydrochloric acid overnight then rinsed three times with distilled water and a further three times with milli-q water). 80 mL of unfiltered water was collected in a similar way. Samples were kept on ice and then frozen. Filtered samples were thawed then analysed colourmetrically for DIN (assumed from combined nitrate-nitrite due to low nitrate concentrations). SRP using an automated, high throughput, water chemistry machine, the Easychem Plus (Systea, Italy). All chemicals used to make standards and reagents were of reagent grade and methods used by machine standard. The unfiltered water samples were thawed then processed for total phosphorus (TP) using standard colourmetric methods and testing for absorbance on a Trilogy Laboratory Fluorometer using a PO4 module.	American Public Health Association (APHA), 1995	
Soluble reactive phosphorus (SRP)	µg/L			
Total phosphorus (TP)	µg/L		Wetzel et al. 2000	
Algal cover*	%	Visual estimate of % of the stream bed covered by algae.		Predictor variables
Macrophyte cover*	%	Visual estimation of % of the stream bed covered by macrophytes.	Harding et al. 2009	
Shading*	%	Visual estimate of % of the stream bed shaded when the sun is overhead.		
Discharge	m^3/s	Discharge measured across one run, evenly flowing and free of obstructions. Offsets were placed wherever depth or discharge changed noticeably, with no fewer than five per transect. Water depth read on the downstream side of a ruler and water velocity is measured four-tenths of water depth up from the bed using a Marsh-McBirney Flow Mate. Discharge was calculated based on standard methods.	Gordon et al. 2004, Harding et al. 2009	
Wetted Width	m	Wetted width was measured at five locations along the 20m reach.	Harding et al. 2009	

* Variables considered constant, only assessed on the March sampling run.

3.2.4 Land use, riparian conditions and other GIS derived variables

For each site, catchment area was defined using the River Environment Classification (REC, Snelder et al. 2005). Due to the low topography of the Plains, catchments generated from digital elevations only approximate actual catchment areas, so manual changes were made to fit known water movement through ground truthing. This was further complicated by the stock water-races and irrigation canals, which result in considerable cross-catchment transfer of water. To estimate the relative inputs of these artificial waterways between sites, an upstream distance measure which included their length was calculated. The elevation change across the reach that best approximated the location of each site on the REC was calculated (Snelder et al. 2005), while stream order and the number of junctions upstream were counted manually. All parameters related to stream channel distance are based on the manually constructed stream maps. The area of each catchment situated within a 100 m buffer of the stream channel was also calculated. For each catchment and 100 m buffer zone area metrics of proportional land use, forest presence, urban development and road density using ArcMap 9.2 tools, the New Zealand Land Cover Database ver.2 and AGRIBASE (Terralink 2004 Agribase 2009 – Table 3).

Table 3: Distance, land use buffer zone and other site-wise variables derived by GIS, for use as predictors in analyses. Where no units are given the variable is a proportion.

Site-wise Variables	Methodology/calculation (all variables are based on the stream maps I constructed)	Units
Stream distance to most downstream site	The length along the stream channel to the most downstream site (Bramleys Rd)	km
Upstream natural channel length	The length of all natural upstream channels (including all tributaries) based on traced and ground truthed stream maps. The decision was made to "cut off" a natural stream reach (beginning the "unnatural" portion), when upstream reaches took the form of highly channelised, free-flowing canals, of uniform depth, characteristic of water-race or irrigation networks (Google Inc. 2009, ArcMap 9.2).	km
Upstream channel length including water race	The length of stream including all natural and man-made waterways upstream of the site. Where branches split in two in the direction of flow, I only measured the branch connecting to the system. (Waimakariri District Council water race data in geospatial form and ground truthing).	km
Upstream junctions	Counted manually (excluding the water race network).	
Stream order	Counted manually (excluding the water race network).	
Catchment area upstream	Catchment areas were defined using the River Environments Classification (REC), then altered manually to meet known directions of water flow and channels not delineated by this model (Snelder et al. 2005).	km^2
Area of 100-m buffer zone upstream	The area upstream of each site within 100m of the natural stream. This distance was chosen based on its use in buffer zone delineation (Baker et al. 2007) and as the smallest distance class to measure land use metrics to assess critical distances of impact (Houlahan and Findlay, 2004) (ArcMap 9.2).	km^2
Length of road per km^2 of upstream catchment area	The total length of all paved and meteled roads in each upstream catchment area Zealand Landcover Database ver.2, (Terralink 2004), expressed as a fraction of the area in km^2.	
Proportion upstream catchment area that is urban	The proportion of the catchment area that is built-up (New Zealand Landcover Database ver.2, Terralink 2004))	
Proportion upstream buffer zone that is urban	As above but for the 100-m buffer zone.	
Proportion upstream catchment that is in moderate to high intensity farming	The proportion catchment area that is in use as Dairy, Beef, Sheep or Sheep and Beef (Agribase 2009)	
Proportion upstream buffer zone that is in moderate to high intensity farming	As above but for the 100-m buffer zone.	
Proportion upstream in each of each of 6 land use types	The proportion catchment area that is in use as dairy, sheep, beef, sheep and beef, arable and lifestyle as defined by the AGRIBASE (Agribase, 2009)	
Proportion upstream buffer zone in each of each of 6 land use types	As above but for the 100 m near zone.	
Proportion upstream catchment with forest cover	The proportion of the catchment covered in exotic or native forest (New Zealand Landcover Database ver.2, (Terralink 2004))	
Proportion upstream buffer zone with forest cover	As above but for the 100-m buffer zone.	
Elevation change within catchment.	Based on the reach of highest and lowest elevation in all mapped stream reaches that correspond to REC reaches (Snelder et al. 2005).	m
Average level of macrophyte cover of a upstream sites	An average of the percentage cover of macrophytes of all upstream sites (Table 1)	%
Average level of shading cover of a upstream sites	An average of the percentage cover of shading of all upstream sites (Table 1)	%

3.2.5 Analyses

3.2.5.1 Space-time interaction

I tested the influence of the space-time interaction on the composition of water quality in the Cam River to determine if the spatial pattern varied through time and adapted the subsequent analysis approach depending on the presence of a significant interaction (Legendre et al. 2010). A significant space-time interaction would have led to separate analyses for each sampling period, whereas a non-significant interaction meant subsequent analysis was able to be restricted to analysis of one sampling period because the results obtained from different periods should be qualitatively similar (Laliberté et al. 2009).

Canonical redundancy analysis (RDA) can be used as a form of multivariate analysis of variance (MANOVA, manovaRDa) to test the relationship between a response matrix, typically species abundance and two crossed factors (Legendre and Anderson, 1999). In this case, space and time were the two factors and the water chemistry parameters were the equivalent of "species". Orthogonal dummy variables (Draper and Smith 1981), (otherwise known as Helmert contrasts were used to code for the two factors. However, where no replication exists, as in this study, the space-time interaction cannot be tested in the classic two-way ANOVA as no degrees of freedom remain to test the denominator of the F statistic (Legendre et al. 2010). To avoid this problem, spatial and temporal principal coordinates of neighbor matrices (PCNM) can be used to code for the space-time interaction (Legendre et al. 2010). The PCNM method takes into account close neighbourhood relationships among sites by using eigenvalue decomposition of a truncated matrix of geographic distances among the sampling sites. The eigenvectors produced, corresponding to positive eigenvalues, are used as spatial descriptors in regression or canonical analysis (Borcard and Legendre 2002, Dray et al. 2006). The interaction is modelled using variables that are the product of the first s/2 and t/2 spatial and

temporal PCNM variables, respectively (where s is the number of sites and t the number of sampling times) (Laliberté et al. 2009). Empirical simulations show that this model has correct Type 1 error and that its power is equal or greater than other potential ANOVA models (Legendre et al. 2010).

I used a mixed model in which space was considered a random factor and time fixed. Space was considered random because although our site choices conformed to the selection protocol previously mentioned, the path of the river across the landscape itself can be considered random. Time was considered fixed as sampling occurred at regular intervals during the specific period of interest, in which no major disturbance events occurred. A non-significant interaction effect led to the testing of space and time using a basic RDA, without replication, rather than being treated as co-variables, as in the previous analysis (Laliberté et al. 2009). All tests were performed with 999 permutations of the residuals (Anderson and Legendre 1999). The manovaRDa procedure was carried out in the R program (R Development Core Team 2007) using the package "STI", with Model 5 used when the interaction effect was included and Model 2 when without the interaction (Borcard and Legendre. 2002, Dray et al. 2006, Legendre et al. 2010) The "STI" contains functions found in both the 'vegan' and 'PCNM' packages (Dray et al. 2006, Oksanen et al. 2007).

3.2.5.2 Water quality change through time

I used non-multi-dimensional scaling (NMDS), a type of indirect gradient analysis that reduces the dimensionality of multivariate data to aid in the interpretation of water quality in the Cam River (Jongman et al. 1995). NMDS has been shown to be a more robust method of indirect gradient analysis than principal components analysis or detrended correspondence analysis (Minchin, 1987). NMDS reduces the dimensionality of multivariate data by describing major trends among sites by the joint occurrence of similar levels of each parameter or species (Jongman et al. 1995). Much variation is often

captured in two or three derived axes which can then be related graphically or statistically to other variables to reveal patterns (Hawkins et al. 1997).

Euclidean distance is commonly used when conducting all types of gradient analyses on environmental data (Clarke and Corley 2006). However, Euclidean distance is strongly influenced by whether the data are rescaled or standardised in ordination procedures (Jackson 1993). The scaling of variables is important when dealing with environmental data, as they often have different units of measurement meaning that large numbers will completely obscure variation in the small numbers, resulting in a matrix that is dominated by one variable. This problem was avoided by standardising the variables so the each variable was given an equal weight (McCune et al. 2002). I did this using normalisation in PRIMER, which involved subtracting the mean and dividing by the standard deviation for that variable (Clarke and Corley 2006). It sets the mean at 0 and standard deviation at 1 and each variable is adjusted to an equivalent value in the range of approximately -2 to 2. This achieves a local shift and scale change crucial for making such variables comparable with each other (Clarke and Corley 2006). However, this procedure does not solve problems related to non-normality so transformations were done first. Phosphorus variables (SRP and TP) were log-transformed, while DIN and turbidity were square-root-transformed, as these variables were less skewed.

Non-metric multidimensional Scaling (NMDS) using Euclidean distance and 999 permutations was carried out on the transformed set of water quality data for all sites and samples over the three repeats in 2010 and 2009 sampling of the Cam River in Primer (Clarke and Corley 2006). This resulted in a comparable measure of water quality across all sampling runs. Euclidean distance in NMDS space, between sites, across the three sample runs, was calculated and used to measure change in water quality variables between sampling occasions (as in Scarsbrook 2002).

To assess if the change in water quality was greater between any two consecutive sampling runs compared with any other two consecutive runs, paired-sample T tests were performed on the change in

NMDS space values between corresponding sampling occasions (Zar 1999, R Development Core Team 2007).

3.2.5.3 Factors relating to change in water quality composition

I examined the relationship between the amount of change in water quality that occurred at sites between the January and February sampling events and the size of sites using Quantile regression. The plot of change of NMDS space against the log of each size variable gave a better spread to the data with a more observably linear decrease in NMDS change, so was used in analysis. Quantile regression was performed to fit regression lines that bound the upper 10 percent of the data in the R package quantreg using the "br" method, which is a variant of the Barrodale and Roberts (1974) simplex algorithm and is suitable for data sets of this size (R Development Core Team 2007, Koenker 2009). These were tested for significance using the "boot" method to compute standard errors (Koenker 2009).

To determine which landscape level factors influenced the amount of change in water quality composition through time I used a model selection approach in the program Spatial Analysis in Macroecology (SAM, Rangel et al. 2006). This procedure uses Akaike's Information Criterion (AIC_c) to select the best model out of many competing models providing a parsimonious balance between model complexity and predictive power (Rangel et al. 2006, Rangel et al. 2010). SAM evaluates all models that emerge from a set of explanatory variables and ranks them according to their AIC_c value and derived statistics (e.g. Akaike's weight and delta AIC_c). A single predictive "multi model" can be generated by averaging and weighting the estimated model parameters as a function of Akaike weights (Rangel et al. 2010). Although this model is based on ordinary least squares regression, the problem of inflated type one errors (due to spatial autocorrelation in the residuals) can be tackled by adding spatial covariates as "fixed" predictors in the model selection procedure (Diniz-Filho et al. 2008). I used "stream distance to most downstream site" as a fixed predictor in the model selection process as many

other predictors were related to flow and this measure is more meaningful than Euclidean distance in the case of streams (Lyon et al. 2008). Results are displayed for models with a change in AIC_c (ΔAIC_c) of less than 2, as these are statistically equivalent to the minimum AIC_c model (Rangel et al. 2006).

3.2.5.4 The impact of tributaries on longitudinal water quality variation

To initially explore system-wide patterns in water quality, I displayed each parameter visually using graduated colour symbols to represent the value of each site, at its actual location. I studied longitudinal changes in the instantaneous flux of nutrients (total phosphorus, soluble reactive phosphorus and DIN) along the Northbrook, the longest branch of the Cam River, using a mass balance approach. Based on the results of the space-time interaction test, analysis of the spatial patterns analysis of water quality from one sampling event is sufficient (Laliberté et al. 2008) Thus this approach was restricted to the March snapshot sampling event as this was the most intensive.

Total instantaneous loads of each nutrient species were calculated for each site along the mainstem of the Northbrook and the site closest to the confluence on each tributary, by multiplying the concentration by the discharge at that site and time (similar to the method of Grayson et al. 1997). It is common in water quality studies for point-sources total loads (tributary and drain inputs in this case) to be subtracted from the total loads at main stem sites in a longitudinal manner (Grayson et al. 1997, Salvia et al. 1999, Behrendt and Opitz 1999). If the remainder is a positive number, the increase is attributed to non-point source inputs along the intervening reach (Grayson et al. 1997), whereas a negative number is attributed to in-stream uptake by biota or sedimentation (Finlay et al. 2011, Salvia et al. 1999). This mass balance approach was used to calculate expected values of total and dissolved phosphorus, DIN for sites along the Northbrook branch, and then compared to the actual instantaneous flux. I calculated the deviation difference between observed and expected values at each site along the main stem of the Northbrook. I also plotted the observed and expected flux of each nutrient in relation

to the distance of sites from the source of the Northbrook. The flux of each nutrient at tributary sites was superimposed at the distance at which they entered the mainstem, to allow estimation of their influence.

3.3 RESULTS

3.3.1 Space-time interaction

The space-time interaction on water quality was not significant when comparing between the 2010 sampling events or between the March 2009 and March 2010 events (P =0.40 and 0.63, respectively, Table 4). Thus the spatial patterns of water quality were stable through time and temporal variations were common to all sites. Tests of the main effects of space and time showed that water quality in the Cam River was strongly spatially structured for both combinations of sampling events tested (P =0.001); this was true irrespective of the exclusion or inclusion of the interaction term in the model for the test of 2010 events (Table 4). By comparison, water quality did not vary significantly in time, in either combination of sampling events considered, when the interaction term was excluded from the model (P = 0.8 and 0.27 respectively, Table 4). This means that there were few temporal changes in water quality over the 2010 summer and between the months of March in each summer.

Table 4: Effects of space, time and their interaction on the levels of all physico-chemical water quality measures during repeat sampling events of the Cam River. Spatial and temporal principal coordinates of neighbour matrices (PCNM – Borcard and Legendre 2002) were used to code for the space-time interaction in a manova-like RDA procedure (Laliberté et al. 2009). A mixed model was used in which space was considered a random factor and time fixed. A non-significant interaction led to the testing of space and time using a basic RDA, without replication. Analyses were conducted using the "STI" package in the R program (R Development Core Team 2007).

	Three sample events in 2010			March 2009 to March 2010		
	R^2	F^a	P	R^2	F^a	P
Source						
Space x time	_	1.12	0.40	_	0.81	0.63
Interaction excluded in the model						
Space	0.52	2.28	0.001*	0.56	1.29	0.001*
Time	0.008	1.12	0.8	0.011	1.07	0.27*
Interaction included in the model						
Space	0.52	2.35	0.001*	0.56	1.17	0.28
Time	0.008	1.10	0.21	0.011	1.19	0.21

* Values are significant at $P \leq 0.05$. All tests were performed using 999 permutations of the raw data.

[a] Pseudo-F

3.3.2 Water quality changes through time

Eight measured physico-chemical variables were included in the NMDS analysis. Axis one of the NMDS was strongly correlated (Spearman rank correlation $r_s > |0.6|$) with total phosphorus and turbidity (r_s = -0.75 and -0.69 respectively). While axis two was strongly correlated with nitrate (r_s = -0.63) and dissolved oxygen (r_s = -0.69) (R Development Core Team 2007). Many sites remain central and move only a small distance in NMDS ordination space between sampling events, however, there were some sites with more extreme values in each sampling event (Figure 2).

No significant differences were found between the consecutive levels of change in water quality across the four sampling occasions using paired two-sample t tests (January to February compared with February to March, p = 0.095; March 2009 to Jan 2010 compared with January to February 2010, p =

0.097). This suggests that the amount of change in water quality was approximately equal between each sample run.

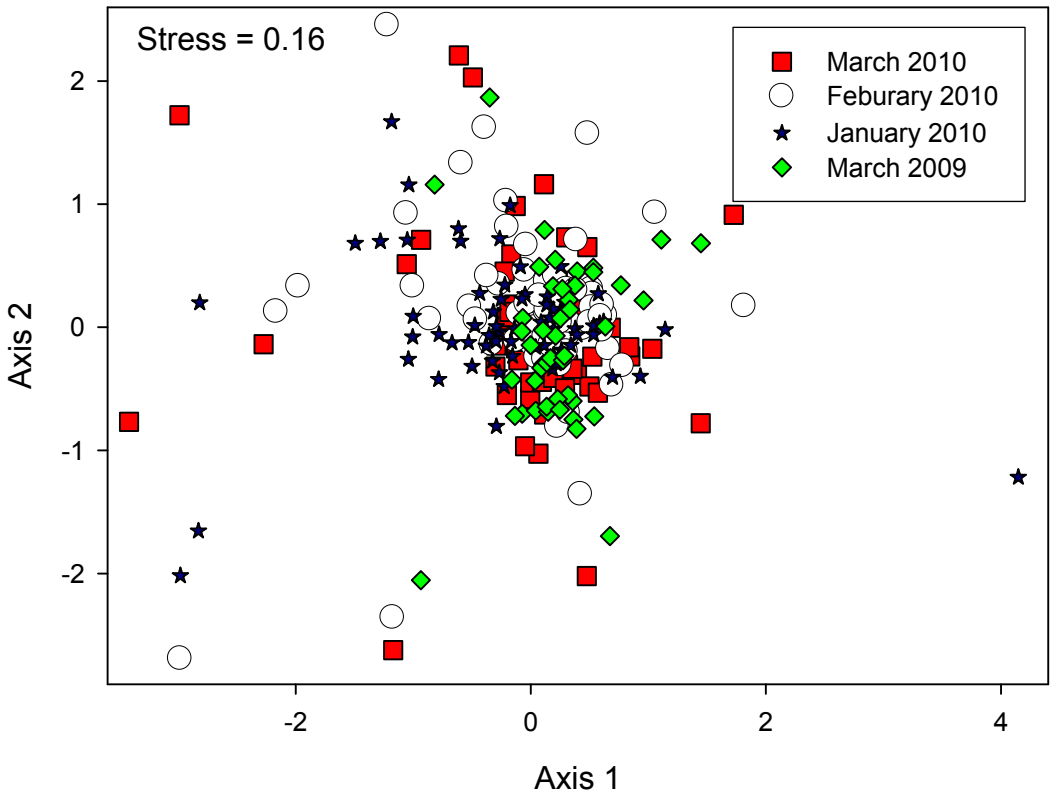


Figure 2: Water quality of sites sampled on the Cam River in each of four sampling runs represented in two dimensional Non-metric multidimensional scaling (NMDS) ordination space. A stress value of less than 0.2 and indicates a good representation of community relationships between sites.

3.3.3 Factors relating to change in water quality composition through time

Sites on smaller sections of the river, lower in discharge and closer to headwaters, had greater changes in water quality (Between the January and February sampling events) than larger sites further down the network (Figure 3). Smaller sites were also more variable in terms of the amount of change in NMDS space that occurred between sampling runs (Figure 3). The upper level of variation in the temporal change in water quality significantly reduced with increasing discharge and length of upstream natural

channel ($p = 0.004$ and $p \leq 0.001$, Figure 3. a, and b, respectively). The temporal variation of water quality in small streams was highly variable when compared with the homogeneous nature of larger streams.

The best model (determined by Akaike's Information Criterion, AIC_c), explained 20% of the variance in the relationship between catchment, buffer and in-stream variables and the amount of change of water quality between sampling events. The variables stream order, catchment area upstream and the proportion of the catchment area in dairy and forested were included in the best model (Table 4). However 17 models performed equally well ($\Delta AIC_c < 2$) and included combinations of 14 variables, (including stream distance to lowest site, which was held constant) of the many which were put forward for model selection (All from Table 3 and predictors from Table 2). The four most important variables (averaged over all models) explaining change in water quality were stream order, the proportion of the catchment area in dairy and forested, and upstream channel length including water-race (listed in order of decreasing importance, Table 5). The change in water quality levels was greater in reaches that were smaller and further up in the system, and also greater in those with a higher proportion of catchment area that was forested and a lower proportion as dairy farm (relative to other sites on the network) (Coefficients, Table 5)

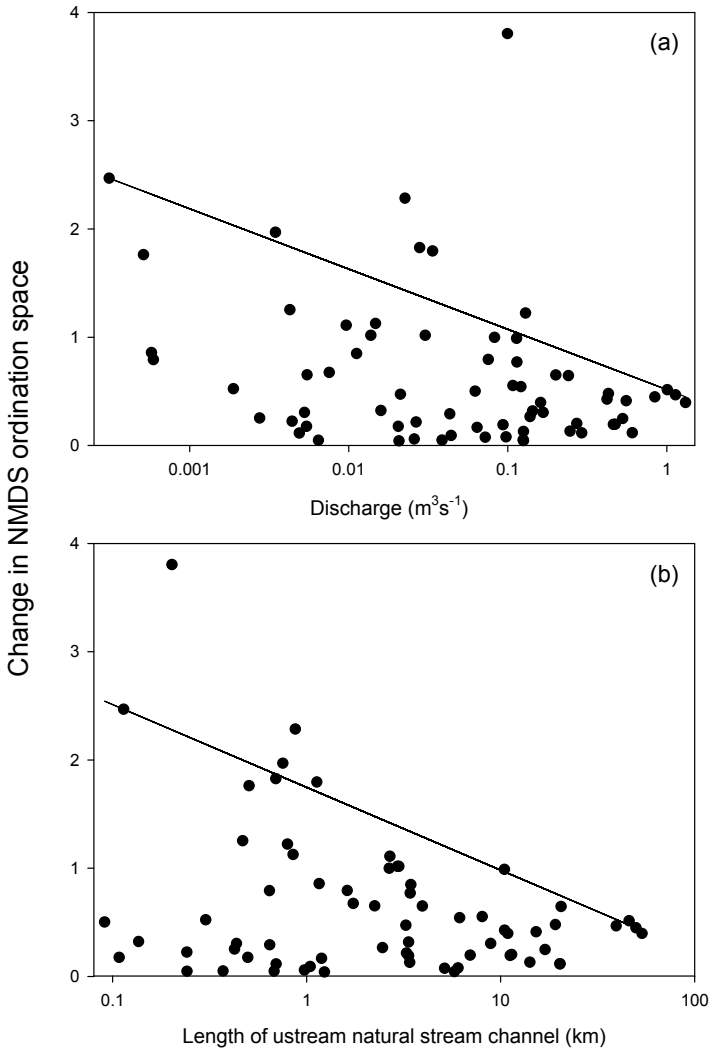


Figure 3: Change in Non-metric multidimensional scaling (NMDS) ordination space between sites in the January and February sampling events against the average discharge across both sampling events (a) and the length of natural stream channel upstream of a site (b). The upper 90th Quantile regression lines are shown on each. Note the log scale on the x axis of both graphs.

Table 4: Parameter estimates for variables selected in the best Ordinary Least Squares model according to the Akaike Information Criterion (AIC_c) after testing combinations of all possible models. This model explained the most variance, as determined by the R^2 and AIC_c, in the amount of change in water quality (community of physico-chemical variables), between the 2010 January and February sampling events in the Cam River. The response variable was the average change in Euclidean distance in the water quality Non-metric Multi Dimensional Scaling (NMDS) ordination space. The predictor variables were the suite of catchment, buffer and in-stream variables identified as predictors in Table 1 and all those in Table 2, in addition to the spatial autocorrelation variable held fixed. AIC_c indicates the fit of each model, taking into account the number of variables to avoid over fitting.

Variable	**Coefficient**	**Standard Error**
Constant	1.82	0.38
Stream order	-0.45	0.12
Catchment area upstream	<0.001	<0.001
Proportion of upstream catchment area in dairy	-0.96	0.38
Proportion of upstream catchment area forested	5.92	2.98
Stream distance to most downstream site	-0.073	0.046

Note that "Stream channel distance to lowest site" is included in every model to account for the spatial location of each site.

Table 5: The importance (the number of times the variable was selected, averaged over all models) of catchment, buffer and in-stream variables predicting change in water quality composition between the January and February sampling events in 2010, using the Akaike Information Criterion (AIC_c) model selection approach. Variables reported were chosen at least once in models that had a change in AIC_c (ΔAIC_c) of less than 2, indicating that they are not significantly different in their ability to predict change in the water quality composition. The number of time each variable was selected in these models was indicated. The spatial variable, "stream distance to most downstream site" had an importance of one as it was included in the selection as a fixed predictor, thus was forced to be included in every model. The coefficients are shown to indicate the direction of the relationships.

Variable	Importance	No. times selected	Coefficient	Standard Error
Stream order	0.78	16	-0.37	0.13
Proportion of upstream catchment area in dairy	0.70	14	-1.08	0.40
Proportion of upstream catchment area forested	0.57	14	5.89	1.99
Upstream channel length including water race	0.47	2	-0.007	0.002
Proportion of upstream catchment area in sheep	0.46	2	3.46	1.24
Proportion of upstream catchment area lifestyle	0.43	2	-2.04	0.68
Catchment area upstream	0.39	5	<0.001	<0.001
Proportion of upstream catchment area urban	0.37	6	-0.50	0.22
Area of 100 m buffer zone upstream	0.31	2	<0.001	<0.001
Proportion of upstream catchment area in beef	0.30	1	-1.61	0.58
Wetted width	0.25	2	0.06	0.03
Upstream natural channel length	0.25	1	0.004	0.006
Average discharge	0.24	1	0.29	0.18
Stream distance to most downstream site	1.00	17	-0.05	0.06

3.3.4 The impact of tributaries on longitudinal water quality variation

Sites along the Northbrook branch of the Cam River system cover a wide range of each of the nutrient parameters measured (Figure 4, Table 6). The instantaneous flux of total phosphorus (TP), soluble relative phosphorus (SRP) and dissolved nitrate-nitrite (DIN) increased with increasing stream channel distance from the source of flow (Table 4, Figure 6). The flux in the lower reaches of the Cam River was greater than that in the upper Northbrook by one order of magnitude for TP and SRP and two orders of magnitude for DIN. Sharp changes in the observed fluxes were predicted well by the expected

fluxes when responding to the entry of tributaries. The magnitude of these changes was generally proportional to the flux of nutrients in incoming tributaries (Figure 6).

The flux of TP and SRP was less than expected in the lower reaches of the Cam River and began to decline from approximately 5-6 km downstream from the source of flow. At this point the large tributaries, the Southbrook and Coldstream had combined with the Northbrook to form the main Cam River. There were mismatches between the observed and expected values through the mid-section of the Northbrook in the flux of both TP and SRP, the expected values variously over and under estimated the observed levels, with a pattern that was not consistent between the forms of phosphorus. The observed flux of DIN matched the expected flux more closely than those of TP and SRP. At 5-6 km downstream of the source, following the confluence with the Southbrook and Coldstream, the flux of DIN in the Cam River began to decrease, however this decrease was only gradual compared with a steep decline in the expected values of DIN flux. This resulted in the expected values of DIN flux underestimating the observed levels in the lower reaches of the Cam River.

Table 6: Mass balance of the instantaneous flux of total phosphorus, phosphorus and dissolve nitrate and nitrogen (DIN) at sites along the Northbrook branch and then mainstem of the Cam River during the March sampling event in 2010. Each row contains data from one site, sites become further from the headwaters moving down the table, rows containing mainstem sites are shaded. The “Mainstem” and “Tributary” columns give the flux based on water sample nutrient concentrations and discharge of this sampling event, at all sites along the main branch of the river and at the site on each tributary closest to the confluence with the mainstem, respectively. The values in the “Expected mainstem” columns represent the flux expected at a site based on the flux at the site directly upstream and with the addition of any tributary flux(es) that have entered. The “Diff from expected” column gives the difference between observed and expected flux at each mainstem site, the four largest differences in each case are in bold

Sites	Instantaneous total phosphorus flux (ug/s)				Instantaneous SRP flux (ug/s)				Instantaneous DIN flux (mg/s)			
Downstream Distance (km)	Mainstem	Tributary	Expected mainstem	Diff from expected	Mainstem	Tributary	Expected flux	Diff from expected	Mainstem	Tributary	Expected flux	Diff from expected
0.00	0.68				0.25				0.0001			
0.30		0.14				0.10				0.0001		
1.04	0.54		0.82	-0.28	0.58		0.26	0.32	0.0007		0.0002	0.0005
1.22	0.08		0.54	-0.46	0.08		0.58	-0.5	0.0001		0.0007	-0.0006
1.96	1.94		0.08	1.86	2.20		0.08	2.12	0.0042		0.0001	0.0041
2.13		0.02				0.02				0.0001		
2.14	0.81		1.96	-1.15	0.73		2.2	-1.47	0.019		0.0043	0.015
2.39	2.52		0.81	1.71	1.36		0.73	0.62	0.032		0.019	0.013
2.40		0.53				0.33				0.013		
2.64	2.81		3.05	-0.24	2.29		1.69	0.6	0.063		0.045	0.018
3.09	2.81		2.81	0	2.31		2.29	0.02	0.056		0.063	-0.007
4.02		1.64				2.01				0.061		
4.09	2.17		4.49	-2.28	1.41		4.32	-2.91	0.027		0.12	-0.09
4.12	2.37		2.17	0.2	4.63		1.41	**3.22**	0.10		0.027	**0.073**
4.12		4.48				2.60				0.074		
4.13	5.56		6.85	-1.29	7.57		7.23	0.34	0.16		0.17	-0.014
4.18		0.16				0.17				0.0004		
4.19		0.09				0.02				0.0014		
4.72		0.05				0.03				0.0005		
4.88	6.21		5.85	0.35	0.16		7.79	**-7.63**	0.13		0.16	-0.032
4.94	11.58		6.21	5.37	4.67		0.16	**4.51**	0.17		0.13	0.04
4.95	7.61		11.58	-3.97	3.45		4.67	-1.22	0.14		0.17	-0.03
5.20	7.21		7.61	-0.4	9.35		3.45	**5.9**	0.20		0.14	0.06
5.55		4.57				2.64				0.14		
5.57	11.73		11.78	-0.06	8.47		11.99	-3.52	0.60		0.34	**0.26**
6.10		4.10				1.11				0.065		
6.44	18.54		13.87	4.67	8.55		9.58	-1.03	0.60		0.67	**-0.065**
6.45		0.23				0.06				0.01		
7.20	15.03		18.78	-3.75	6.24		8.61	-2.37	0.36		0.61	**-0.25**
7.73		0.45				0.06				0.078		
8.24	13.50		15.49	-1.98	6.79		6.3	0.49	0.45		0.44	0.012

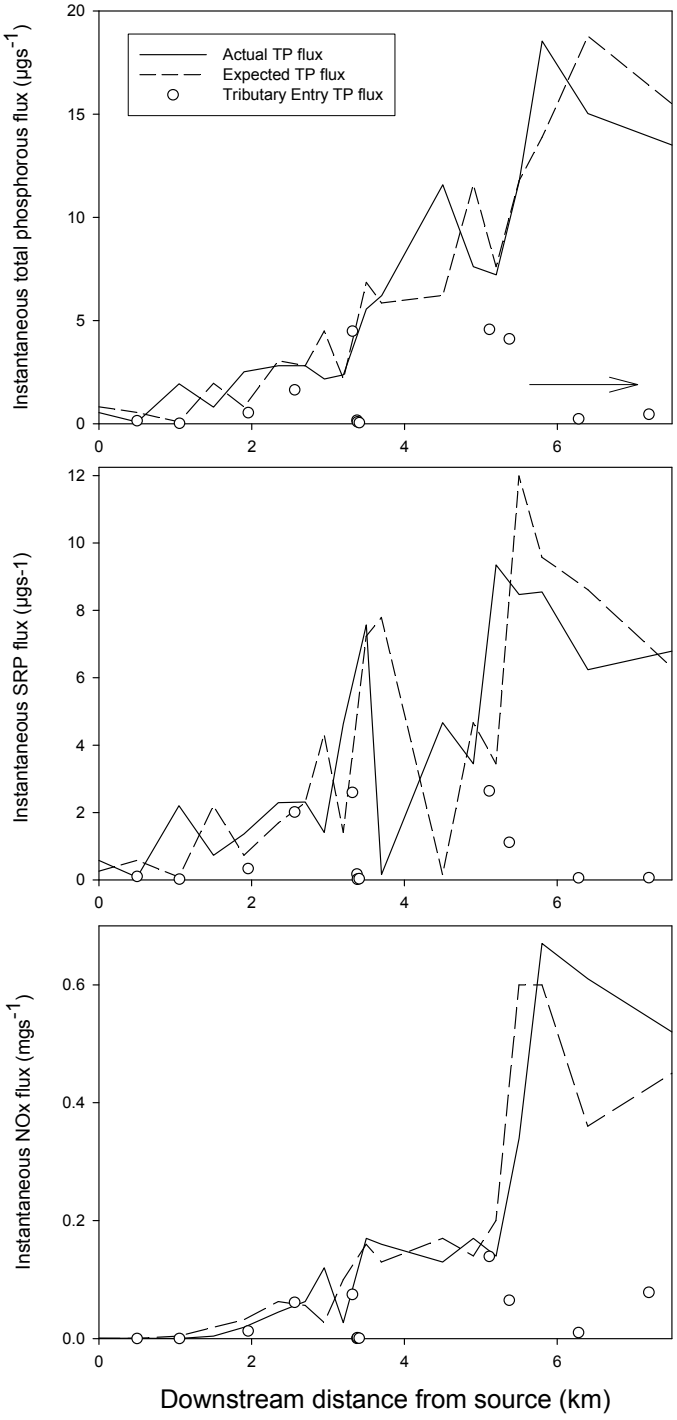


Figure 4: Longitudinal profiles of the actual and expected water quality along the Northbrook branch of the Cam River during the March 2010 sampling event, based on the calculations presented in Table 6. The values at sites are plotted against the stream channel distance from the source of the point of this branch (sites become further from the source moving along the x axis). The flux of each tributary entering the mainstem is plotted against the distance from its confluence to the source (however actual locations of these sites were 25-50 m upstream on the tributary). The key in (a) applies to all panels. The arrow represents the direction of downstream flow through mainstem sites.

3.4 DISCUSSION

The water quality of the Cam River displayed a high level of spatial variability and small stream sites were more temporally varied than larger sites. Yet overall, the spatial pattern of water quality was consistent through time. Together these findings provide an affirmation of the merits of catchment wide snapshot sampling events, combined with spatial analysis, to best characterise the water quality of a river system.

3.4.1 Consistent spatial water quality patterns through time

My results indicate the spatial patterns of water quality were stable through time and temporal variations were common to all sites. The temporal change in water quality was approximately equal between each sample run across all sites. Overall, relatively little change occurred in water quality between sampling runs, across months and between years, at least within the period of the late summer season sampled at base flow. This conforms to the theory that base-flow water chemistry is relatively stable and sampling during this state of flow avoids the problem of temporal bias in large scale studies (Biggs et al. 1990, Pionke et al. 1999, Lyon et al. 2008).

Although at some sites, the level of individual parameters changed between sampling events, taken as a multivariate community of physical and chemical species, the overall quality of the water was stable at a large number of sites. Small changes that occurred were of a comparable magnitude and direction across all sites, thus the water quality of sites relative to each other remained similar, retaining a consistent system-wide spatial pattern. Temporal variability in stream water quality has been associated with flow rate which can change the biochemical properties, particularly during floods (Dahm et al. 1998, McKee et al. 2001, Scarsbrook 2002). Despite discharge variation of up to 0.5 m^3s^{-1} between sampling events at the most downstream sites sampled, the relatively constant

spatial pattern of water chemistry in the Cam River system suggests that any smaller changes in flow that may influence water quality are occurring at least uniformly across the catchment.

A similar pattern was observed in an early spatially intensive study that linked conductivity to subcatchment geology (Walling and Web 1975). In that study, under high flow conditions the range of conductivity values was reduced, but the general spatial pattern stayed the same (Walling and Web 1975). Yet in other studies, temporal stability in concentrations of dissolved nitrogen (Pionke et al. 1996), phosphorus (Pionke et al. 1996, Salvia et al. 1999,) and other pollutants such as sodium and chloride (Wayland et al. 2003) has been observed between seasons and across years, despite changes in flow.

Contrary to the pattern I found in the Cam River, certain nutrients can vary temporally in a complex way under base flow conditions (Clow et al. 1996, McKee et al. 2001, Wayland et al. 2003). For example, nitrogen can exhibit both temporal stability (Pionke et al. 1996) and variability (Valett et al. 1996, Salvia et al. 1999). Nitrogen levels in stream water can decrease during low flow periods through increased nutrient uptake by algae, macrophytes and microbes (Valett et al. 1996). This is the result of increased growth rate under warm, low flow conditions, combined with longer contact time for absorption (Valett et al. 1996). Thus, it has been suggested that a single snapshot sampling event cannot adequately describe the variability of a stream (Wayland et al. 2003). However, the consistent base flow spatial pattern of water quality in the Cam River suggests a clear picture of water quality throughout the system can achieved with a single sampling event. This picture would include an indication of which parts of the system had low or healthy water quality, relative to each other, where extreme values are more likely to be observed, and an idea of the average condition of the system. For example a snapshot water quality study in Australia found large discontinuities in the balance of total phosphorus along a main stem, this was later equated to an unlicensed point source discharge from fish farm (Grayson et al. 1997. This information was conveyed to an environmental agency and allowed mitigative action to take place.

The drawbacks of the snapshot sampling approach, which include logistics and cost of personnel, equipment and fuel, as multiple teams often need to work the simultaneously, have led to it receiving limited attention in literature and limited use in water quality monitoring (Grayson et al. 1997, Eyre and Pepperell 1999). The finding that one snapshot is enough in systems like the Cam River has the potential to encourage managers and scientists to use this approach. Care must be taken in the application of information from snapshot sampling; it cannot be scaled up to yearly budgets due to the disproportionate export of certain nutrients during floods (Eyre and Pepperell 1999, McKee et al. 2001). However, as base flow conditions exist for 10 to 11 months of the year in New Zealand (Biggs et al. 1990), the conditions indicated by one snapshot event are likely to be those experienced by the biota which inhabit during a large part of the year, and thus could be used to model habitat availability and species distributions.

I used multiple physical and chemical measures to build a picture of the overall condition of water quality at each site. In this way, small changes in one parameter alone did not offset the spatial pattern in the overall quality of the water. This multi-metric approach allowed a more comprehensive representation of conditions, without losing details. Multivariate studies of the spatial pattern of stream invertebrates (Scarsbrook 2002) and forest tree species (Laliberté et al. 2008) have also found consistent spatio-temporal patterns. In these studies, as in mine, examining the spatio-temporal pattern of each species would be far too complex a task and difficult to relate to environmental factors. The multivariate water quality approach has applicability to modelling the suitability of stream conditions in general, for the community of invertebrates or other biota that may inhabit it (Scarsbrook 2002**).** However, for studies relating to individual species, for which extreme tolerance levels of particular physical and chemical conditions exists, a specific sampling regime is more suitable (as in Stewart-Koster et al. 2007).

A temporal change in spatial pattern of invertebrate and plant communities has been linked to changes in the processes governing temporal variation (Legendre et al. 2010). Different

paradigms of metacommunity structure may exist spatially and temporally, and temporal changes may occur to variable degrees at different sites (Laliberté et al. 2008, Brown and Swan 2010). In light of these ideas, the lack of space-time interaction in the Cam River indicates that the same processes are governing spatial variation throughout the base flow of late summer in both 2009 and 2010.

One process governing water chemistry is external inputs to the system, and these may vary with changes in land use practices and season (Jones et al. 2004). The consistent spatial pattern between March 2009 and 2010 suggests that land use patterns have not changed. However, impacts of land-use change can take some time to appear in water quality and the temporal scale of the present study was potentially too small for any changes in land use practices to have an effect, specifically any occurring in the summer of 2010. There is evidence that many slight changes in farming, such as no-tillage practices, have little impact on nitrogen export (Chaplot et al. 2004). Rates of stocking and fertilizer application were potentially variable during this study, yet no influence was seen in the water quality pattern.

In-stream processes can also govern water-chemistry, as laid out by the nutrient spiralling hypothesis (Newbold et al. 1982). Changes in nutrient patterns can occur in relation to successional time (after a disturbance event) (Dent and Grimm 1999). The lack of disturbance events during the 2010 sampling period potentially led to the constant spatial-temporal pattern in the Cam River.

Variation in connectivity to groundwater is a further process which can cause temporal changes in water quality and may also change differently throughout a catchment (Dahm et al. 1998). Sources of flow may differ throughout the year; in winter, under fully recharged hydraulic conditions, flow often originates from multiple substrata across the whole watershed; in summer, flow can be limited to that from deep substrata (Pionke et al. 1999). The surface-water/groundwater interface is an important control point for longitudinal nutrient interactions and fluxes between uplands and streams (Dahm et al. 1998). Thus, temporal groundwater variability influences the

action of this interface. Locations with continual infiltration and ex-filtration, may maintain stable chemical conditions through time, as is potentially the case in the Cam River system, with its open alluvial soils. Many models that suggest changes in spatial pattern through time are based on steep catchments, where the infiltration and ex-filtration rates vary with slope, ground water recharge and precipitation (Clow et al. 1996). However, the Canterbury Plains which are flat, have little rainfall variation, and often strong connections to aquifers throughout the year, do not fit these models. The consistent pattern of water quality in the Cam River, in spite of flow variation, suggests that perhaps models of temporal water quality variation need to be developed more specifically to meet the conditions of lowland, spring-fed systems.

Finally, many papers dealing with "space-time" are using these terms in a general way and provide summaries of information rather than detailed analyses (Fortin and Dale 2005). In an early study of the temporal variation in nutrient and microbial levels along a series of sewage lakes, Legendre identified the difficulty of such analysis due to a lack of suitable models (Legendre et al. 1984). Thus, using basic models, older studies did not take spatial or temporal autocorrelation into account, which may have had confounding effects (Fortin and Dale 2005, Rangel et al. 2006). The analysis of the space-time interaction using the PCNM approach is unique in its ability to explicitly take space into account and test the interaction term for significance in the absence of replication, not achievable with traditional statistics (Laliberté et al. 2008). This has made the task of analysis of spatial-temporal changes much less challenging.

3.4.2 Temporal variation related to stream size

Although, when the system was considered as a whole, temporal water quality in the Cam River changed little, certain sites were much more variable than others. Sites on smaller sections of the river, lower in discharge, width and closer to headwaters, had significantly larger changes in water quality between sampling events than larger sites further down the network. Decreasing temporal

variance was observed with all measures of stream size, however, stream order was the strongest predictor under the model selection procedure. The dendritic nature of streams likely explains this effect. Small streams make up a disproportionate amount of the total channel length in a system, branching out in all directions on the landscape, therefore making contact with more variable landscapes than the higher orders (Fagan 2002, Bishop et al. 2008, Brown and Swan 2010). The different types of land use are likely to have different amounts of temporal variation to their impacts on stream water quality. For example, some land uses vary considerably in their impacts throughout a year, whereas others are more consistent.

Many tributaries of the Cam River arise from springs, for which flow and condition of water is likely to reflect changes occurring in the wider catchment, and even be controlled by water levels in adjoining catchments, such as the Ashley. Thus, the smaller reaches of the Cam River are likely to respond more directly than mainstems, to changes in the flow and pollution levels of the groundwater of the upper catchments and surrounding areas. Under hierarchy theory, processes of interest are constrained by processes occurring at the next hierarchical level (O'Neill et al. 1989). Temporal variations felt in low order reaches of the Cam River may have been dampened or diluted in effect by confluences with water of differing properties before reaching the higher orders.

Another possible explanation for this increased temporal variability in small streams is the high edge to volume ratio, increasing the contact time of water with the land. In this way, smaller sites may experience disturbance events more than large sites (Scarsbrook et al. 2002); for the Cam River in summer months such events may include low flows. Other stream size factors related to the level of temporal variability at sites on the Cam River were the catchment area and channel length, including the water-race network, upstream of a site. However, the coefficients on both these factors were close to zero, indicating that although they had a relatively high level of importance in models predicting temporal variability, they did influence the variability in any particular direction.

There was a strong relationship between the area of a catchment in use as dairy farm and a decrease in temporal variability. In contrast, the catchment area upstream of a site that was forested, was strongly related to increased temporal variation. This suggests that forested areas have a more variable influence over water quality than dairy farms. This is somewhat counter-intuitive as dairy farms are more likely to vary in their level of impact on streams, for example through changes in stocking rate. However, these relationships may be a facet of the arrangement of land use within the Cam River catchment. The higher order reaches of the Cam River flow through a large dairy farm. This dairy farm may be disproportionately influential in the model due to its location adjoining these higher order reaches which have lower temporal variability in the water quality.

Overall, small sites also have a slightly higher proportion of their catchment that is forested due to the presence of several pockets of bush, the parks of Rangiora and plantings on private land in the upper catchment. Yet a large portion of the temporal variance was not explained by models that included any combination of the variables tested. This suggests that it is either stochastic or associated with some unmeasured factors. It is possible that some environmental change occurred differentially across the catchment, resulting in the greater temporal variability of small streams. However, given the sprawling dendritic nature of the Cam River, small streams are spread throughout the catchment, in combination with the constant spatial pattern in water quality observed through time, this seems unlikely.

Smaller sites were also more variable in terms of the amount of change in water quality that occurred. The negatively sloped limit relationship between the change in water quality and stream size indicates that large streams were not likely to undergo large temporal changes in water quality. Small sites, in comparison, had a range in the amount of temporal change that occurred. A similar decrease in the range of temporal variability with increasing stream order occurred in long term water quality monitoring data from the Seine River (Meybeck et al. 1999). Although this system is

much larger than the Cam River, similar principles relating to the nature of small streams may be the cause of this pattern.

Locations with continual infiltration and ex-filtration of groundwater maintain stable chemical conditions through time (Dahm et al. 1998). Potential changes in the porosity of soil moving towards the coast of the plains, towards more constant connection to aquifers, could contribute to the stability of water quality conditions in the mainstems of the Cam River. This has some implications for the location of single monitoring sites, as it indicates that repeat sampling should be focused more intensively on smaller streams to accurately gauge the extremes of conditions. More monitoring sites should be placed higher in networks as these sites are more likely to respond to temporal disturbance such as intensifying land-use practices.

3.4.3 Unravelling processes influencing spatial water quality patterns

Many of the processes I have discussed relating to temporal variability are also linked to variation in the absolute spatial patterns of water quality in the Cam River. Sites throughout the Cam River system cover a wide range in many physico-chemical parameters measured. Spatial patterns of individual nutrient concentrations and level of physical parameters throughout the Cam River system rapidly build into a complex picture of water quality. As the spatial pattern was temporally consistent, analysis of one sampling event is sufficient to identify processes influencing spatial variation of water quality in the Cam River (Laliberté et al. 2008). I examined spatial variation using data from the March sampling event as this event had the most sites and sites were thus spread the most evenly across the catchment, widening my ability to detect fine scale spatial changes in water quality.

In systems such as streams, where the flow of matter or organisms is directional, longitudinal profiles are a useful tool to tease apart complex variation that exists in the system as a whole. Following the longest branch of the Cam River, the Northbrook, temperature and total phosphorus

concentration both decline, whereas the concentration of dissolved DIN increases downstream and levels of dissolved oxygen varies erratically. A longitudinal increase in temperature is common in streams through widening decreasing shading (Vannote et al. 1980), and patterns of dissolved oxygen have been associated with flow (Salvia et al. 1999). However, in this part of the analysis, I focus on the flux of total and dissolved phosphorus (TP and SRP) and dissolved nitrate and nitrite (DIN), as these are widely accepted as being pollutants of importance in fresh water systems (Hem 1985).

As the discharge through each point in a river system varies, concentrations of nutrients are not an accurate indication of the actual flux of nutrients at any instant. By calculating instantaneous flux in key nutrients, a load balance was calculated for the Northbrook and mainstem of the Cam River which the Northbrook flows into. This allowed point and non-point source inputs to be estimated.

The instantaneous flux presented in this study is proportional to loads calculated during base flow, assuming constant water chemistry. This is a simplified version of mass balance approaches that include measures of ground water and precipitation (Billen and Garnier 2000), yet still allows the identification of processes that are likely to govern water chemistry at various points in the system based on the difference between the measured flux and that expected under an additive model which assumed the entrance of tributaries was the only influence on mainstem water quality and no net retention within the stream (Grayson et al. 1997, Salvia et al. 1999). Variability in the longitudinal levels of each parameter was associated with tributary confluence. Some sharp vertical changes in the profile of nutrient flux at sites along the Northbrook correspond to the entrance of tributaries with correspondingly higher or lower levels of each parameter.

The instantaneous load of all nutrients measured increased down the main stem of the Cam River. However, the observed flux of total and dissolved phosphorus was much less than expected at lower end of the system. A longitudinal decline in the flux of total phosphorus, 60% greater than

expected through dilution by tributaries in a large Luxembourg river catchment, was attributed to in-stream uptake and retention by plants and algae (Salvia et al. 1999). A similar explanation is possible here. I observed an increase in algal levels downstream, along with the increase in suitable substrate and higher light levels that accompanied the widening of the stream. This longitudinal increase in in-stream production was included in early in-stream ecology theory in the River Continuum Concept and is still commonly reported (Vannote et al. 1980, Bott et al. 1985, Finlay et al. 2011).

Observed levels of TP follow expected levels more closely than do those of SRP. SRP has highly variable longitudinal levels; thus the expected level varies between over and under estimating actual load. This is contrary to the findings in the Luxembourg catchment where less difference was seen between observed and expected SRP levels than for TP (mentioned above); and in-stream retention of TP was not as high (Salvia et al. 1999). In several reaches of the Cam River, retention of SRP is higher than that of TP, based on the load balance calculated. This seems logical when considering the preferable uptake of dissolved forms of nutrients by algae and macrophytes (Mulholland 1992).

The discrepancy between the observed profiles of TP and SRP can be attributed to variable external inputs as well as variable uptake rates. For phosphorus, surface runoff is the dominant transfer pathway into the stream. Its affinity for soil particles means it often enters stream largely as TP, which is converted to other forms by in-stream processes (Peterjohn and Correll 1984, Hem 1985). The lower section of the Northbrook (approximately 3.5 to 5 km downstream of the source) had no external tributary inputs until it combined with the Southbrook and Coldstream branches at the main confluence of Cam River. Yet along this segment of the river there are mismatches between observed and expected loads of TP and SRP. At approximately 4.5 km from the source observed TP and SRP are both greater than expected and, in general, remain this way for the next 1-1.5 km downstream. This suggests an external source of phosphorus - at this point the river passes

through a large dairy farm, a land-use type which has been linked to increased sedimentation and phosphorus level (McFarland and Hauck 1999).

Overall, the longitudinal DIN flux matches that expected under the additive load balance approach. This suggests that most nitrogen flowing through (at least the upper part of) the system was already present in the tributaries at the closest sites I measured to the headwaters. As I measured all sites within 500 m of the source of flow, the upwelling of nutrient rich ground water is likely responsible for the residual moderate level of DIN in the upper tributaries of the Cam River.

A gradual increase in DIN flux occurs downstream through the additive coalescence of multiple tributaries of variable DIN levels and flow. The spike in observed values compared with expected at 3 km downstream from the source may represent an unmeasured external input. The Northbrook flows through Rangiora at this point, therefore this input is likely to be an urban storm water drain which I did not locate and measure. The large increases in DIN levels coincide with the entry of the Southbrook, a large tributary of the Cam River which is high in nitrate.

The large increase in DIN, above expected levels in the lower system, suggests some form of disruption to the additive continua has occurred. Nitrogen compounds tend to be associated with groundwater to a greater extent than those containing phosphorus, thus often entering streams as non-point source diffusion or in-stream upwelling zones (Hem 1985, Jones et al. 2001). Downstream increases in DIN have been attributed to both disturbance of sediments in the degraded banks (Grayson et al. 1997) and increased biological activity (Grayson et al. 1997, Finlay et al. 2011).

The sudden increase in DIN at 6 km downstream (above that expected by the confluence with the large tributaries) may be the result of increased groundwater upwelling. The coalescing of the three major branches of the Cam River at this point suggests the course of the river is governed by landscape scale variation which may also influence the interaction of surface and ground waters;

potentially the combining of multiple aquifers or groundwater catchments, which may force groundwater rich in DIN into the Cam River (Dahm et al. 1998, Benda et al. 2004).

The Cam River is situated on an unconfined aquifer, however, as it flows towards the sea it nears a coastal confined aquifer, this also has the potential to increase groundwater upwelling in the lower reaches. However, studies of other rivers have attributed longitudinal increases in nitrate to in-stream sources. For example, in a study in the Eel River, nitrogen levels increased downstream strongly, despite the pristine condition of this catchment due to the large amount of N fixing algae present (Finlay et al. 2011). However, this is not likely to be the case in the Cam River, as N fixing algae do not dominate lowland rivers in Canterbury, thus the upwelling of groundwater offers more potential as an explanation for the downstream increase in DIN. If groundwater upwelling has a role controlling nitrate distributions to an extent in the Cam River, this could have implications for the flow-on effects of changes in groundwater conditions, levels and qualities in general.

The surprisingly large variation in water chemistry composition throughout the watershed suggests collection of data from one site is not adequate to characterise conditions experienced by biota that inhabit the system as a whole. This affirms the merits of the snapshot method over those that lump the whole drainage basin together in analysis, not considering heterogeneity (Walling and Web 1980). Several other key sites are needed to provide more sensitive data. Yet the choice of such sites could not be made without a prior snapshot sampling event such as this one.

Although a load balance model is simplistic compared with models of nutrient retention which include groundwater and rainfall measurements (for example, Behrendt and Opitz 1999, Billen and Garnier 2000), its use in combination with snapshot sampling of the whole Cam River system identified key locations where nutrient additions or transformation processes occurred. Areas of increased nutrients above expected levels could be targeted with specific mitigation plans such as riparian planting and areas of in-stream retention further investigated for their role and protected to ensure continual functioning. An initial snapshot sampling event of key nutrients or other in-stream

parameters of interest provides an efficient way for managers to locate sites for long term monitoring that will best reflect the range of conditions within a catchment.

CHAPTER FOUR - MULTIPLE STRESSORS AFFECT SYSTEM-WIDE SPATIAL PATTERNS IN STREAM ECOSYSTEM PROCESSES

Young cattle grazing beside the Northbrook branch of the Cam River. The single wire fence fails to restrict access. A riparian buffer of exotics, gum trees and gorse is well established on the upper reach.

4.1 INTRODUCTION

The state of a river can vary along an impact gradient from pristine to severely disturbed (Harding and Winterbourn 1995, Young et al. 2008). The effects of this gradient are felt differently across stream types, regions and depending on which component of the stream is under observation (Biggs et al. 1990). The idea of measuring 'river health' is useful because it is readily interpreted by the general public and evokes societal concern about human impacts on rivers. A wide variety of symptoms or bio-indicators can be measured at a broad range of spatial scales (Bunn et al. 1999). River health monitoring has traditionally made use of structural measurements, focusing on patterns in water quality or community composition of aquatic organisms (Barbour et al. 1999, Boulton 1999). However, functional metrics, which quantify processes occurring within an ecosystem, have become commonly used as more complete assessments of river health and its response to changes in catchment land-use (Udy et al. 2006, Fellows et al. 2006). Such processes can help identify the vigour or resilience of an ecosystem, as well as being a direct measurement of ecosystem services, such as organic matter decomposition and primary production (Young and Huryn 1999, Bartkow & Udy, 2004). Ecosystem processes have direct impacts on consumers, the invertebrates and fish which inhabit streams; and thus are of vital importance to sustaining aquatic biodiversity (Winterbourn 2004).

There are several ways to measure ecosystem processes which vary in usefulness for elucidating relationships to stressors and disturbances, as well as practicality and cost (Young et al. 2008, Udy et al. 2006). Indirect measures of ecosystem function can perform as a surrogate for direct measures, such as the rate in-stream metabolism, which can be complex to determine (Morin et al. 1999). Algal productivity on artificial substrates (Biggs 1989, Fellows et al. 2006), as well as rates of leaf decomposition (Gulis et al. 2006) and microbial activity (Niyogi et al. 2001) perform well as indicators of stream health. Rates of nutrient cycling can be used to indicate catchment

disturbance (Udy et al. 2006). These can be measured directly, as rates of denitrification or uptake length of key nutrients (Udy et al. 2006); or indirectly, using the ratios between concentrations of nitrogen, phosphorus and carbon in water as a summation of nutrient processes occurring (Dodds et al. 1998).

Indirect measures of ecosystem processes do not measure the energy transfer between components of the stream in their entirety at a site or reach, but instead measure rates of productivity at a basal level (Winterbourn et al. 2004). As basal food resources such as algae and detritus are essential food web components, the rate of their production needs to be high enough to sustain the aquatic community (Huryn 1998, Biggs 1995). However, above an optimal level, production of basal resources can pollute or clog streams, as is the case with periphyton blooms (Harris et al. 2007). The rates of various ecosystem processes have been widely studied because of their necessary role in primary production, which supports higher trophic levels of stream biota. In this study I specifically investigated spatial variation in four indirect measures of ecosystem processes; the rates of algal productivity, leaf litter decomposition and microbial respiration on these leaves, as well as the DIN:SRP ratio; all of which are related to the functional process, primary productivity.

Ecosystem processes vary spatially and temporally due to a multitude of different stressors (Young et al. 2008). Light, temperature and the availability of nutrients and oxygen, the basic building blocks of photosynthesis and microbial respiration; have all been found to increase rates of primary production (Biggs and Kilroy 2004, Young et al. 2008). Yet, despite much research, few generalisations on the degree to which various environmental factors influence primary productivity are possible (Biggs and Kilroy 2004). An ideal indicator of stream health would respond predictably to anthropogenic stressors but be relatively insensitive to natural and temporal spatial variation (Norris and Hawkins 2000). This ideal is difficult to meet as sensitive indicators respond to all forms of variation in the environment, and anthropogenic changes are often spatially correlated with landscape variation, making these factors difficult to tease apart (Young et al. 2008, Fortin and Dale

2005). The rates of ecosystem processes vary temporally, particularly across seasons (Fuss and Smock 1996, Bunn et al. 1999). However, in a shorter timeframe (weeks to months), processes such as algae growth or leaf decomposition respond to the range of conditions experienced through time; in this way integrating some of the temporal water quality variability often experienced (Salvia et al. 1999, McKee et al. 2001, Biggs and Kilroy 2004).

Heterogeneity in ecological systems arises from complex interactions among multiple causes that operate under a hierarchy of spatial scales (O'Neill et al. 1989). Thus, selection of appropriate spatial scales for these measures is crucial (Bunn et al. 1999). One must distinguish between direct (proximate) effects on processes, resource availability and abiotic stressors that can be patchy at the reach scale, which are in turn often constrained by indirect (ultimate) effects acting at larger scales, such as climate, geology, land use, hydrology and biotic interactions (Stevenson 1997, Biggs 1995).

Previous research regarding the spatial patterns of ecosystem processes has been conducted on a number of different scales. Many studies have investigated large scale differences due to variable climate, geology and land use, comparing processes across biomes (Bott et al. 1985), regions (Mulholland et al. 1987, Findlay and Sinsabaugh 2006), stream types (Biggs and Close 1989) and catchments (Quinn et al. 1997). Such studies usually choose one site per stream, using streams of stream order and flow regime, to minimise natural variation. Another common approach is to compare impacted sites to reference sites in the same catchment or area (Carlisle and Clements 2005, Ferreira et al. 2006, Gulis et al. 2006, Biggs 1989). Alternatively, experimental enrichment had been used at the reach scale to study changes in ecosystem processes in a more controlled way (Gulis and Suberkropp 2003, Elwood et al. 1981). However, studies that use functional indicators to assess catchment-scale spatial processes are lacking.

In recent years stream ecology has emphasised links between streams and landscapes, as well as their nature as connected networks (Power 2001, Brown and Swan 2010, Thompson and Lake 2009). More studies are emerging at the level of whole catchments, utilising a “riverscape”

perspective, under which streams are investigated as whole ecosystems, strongly influenced by their surrounding landscapes at multiple scales (Wiens 2002, Allan 2004). Given the heterogeneity in both landscape and water quality that exists within catchment areas; even those that are small (Prowse 1984) or in pristine condition (Clow et al. 1996, Finlay et al. 2011); considering sites across a network is important in understanding the factors influencing ecosystem processes (Finlay et al. 2011, Stevenson 1997, Harris 1997).

Spatially intensive sampling can highlight patterns in ecosystem processes throughout a catchment, consistent with emerging paradigms promoting the need for spatial and network perspectives in freshwater ecology (Lyon et al. 2008, Fausch et al. 2002). Large data acquired through spatially intensive sampling have provided the opportunity for cross-correlation with parameters that change across catchments and identified geological, land-use and point source contributions to water quality (Eyre and Pepperell 1999), as well as the separation of niche and neutral processes structuring stream invertebrate communities (Blanchet et al. 2008b). Such a spatial approach has been applied sparingly to studies of ecosystem processes, despite the early inclusion of landscape elements and longitudinal gradients in the River Continuum Concept (RCC) (Vannote et al. 1980). Longitudinal changes in various ecosystem process have been considered in light of the RCC (Mulholland 1992, Tank et al. 1993, Pascoal and Cassio 2004), however, little is known about the specific nature of changes within stream networks and how they affect multiple, interacting elements (Ensign and Doyle 2006, Finlay et al. 2011). Without a spatially intensive approach, any heterogeneity of stream health within a system may be missed.

In all ecological fields, the interest in variability associated with landscape and spatial patterns has developed faster than the appropriate statistical methods (Rangel et al. 2006). Previous research into the spatial patterns of ecosystems processes used traditional statistical methods (Finlay et al. 2011, Swanson and Bachmann 1976), which were not well equipped to deal with the spatial autocorrelation which is inherently present in data from stream networks (Tu and Xia 2008). This

has the potential for erroneous results to become incorporated into literature (Dormann 2007). A new spatial analysis method, principal coordinates of neighbour matrices (PCNM, Borcard and Legendre 2002), shows considerable ability to unravel and quantify the processes that drive spatial variation across multiple scales, in many areas of biology (Dray et al. 2006, Laliberté et al. 2008), including freshwater systems (Brind'Amour et al. 2005, Lacey et al. 2007, Blanchet et al. 2008b). Often, landscape-level data are needed for management, because this is the scale where cumulative effects of impacts are evident (Bunn et al. 1999), yet information on ecosystem processes at this scale is lacking.

In this study I examined the variation in ecosystem processes rates within a single river system, to tease apart the influence of various stressors on stream health. The Cam River is a lowland spring-fed network, on the agricultural plains of the Canterbury Region, in the South Island of New Zealand. With urban sprawl and changes in farming practices, this landscape has become a complex mosaic of various forms of agriculture and urban zones (Winterbourn 2008). Worldwide, the study of within catchment variations of water quality and primary production has tended to take place in relatively pristine catchments (Clow et al. 1996, Finlay et al. 2011). The study of degraded catchments, such as the Cam River provides the opportunity to identify a variety of relationships, which may be applicable to other complex, impacted systems. I used four measures of ecosystem processes in this study - rates of algal production on artificial substrates, degradation of leaves in litter packs and microbial respiration on these leaves, as well as the ratio of dissolved inorganic nitrate (DIN) to soluble reactive phosphorus (SRP), as an indicator of the trophic state of a site. These are indirect measures of ecosystem processes and were chosen because direct measures would have been too time-consuming and costly to sample at the site-wise intensity required to sample a whole network (Young et al. 2008).

I included multiple measures of ecosystem processes for two reasons. Firstly, as indirect measures of processes do not link as conclusively to stream health as direct measures (Fellows et al.

2006), the use of multiple measures provided greater information of stream health throughout this system. The use of multiple indicators can increase the accuracy of detection of the level degradation (Bunn et al. 1999). Secondly, comparing the response of these processes to anthropogenic stressors within the catchment revealed the best measure for use by managers of systems such as the Cam River, as well as which particular impacts caused degradation of each process. Then, these findings could be applied to the creation of a specific system-wide management plan.

More specifically in this study, I investigated the spatial patterns in ecosystem processes in the Cam River. I predicted that the rates of algal production, leaf matter decomposition and microbial respiration on leaves should be positively correlated with each other, due to the similarity in their responses to water quality. Variation in ecosystem processes was analysed in relation to stream size and position within the system. I have previously found higher variability in water quality associated with small streams, further up each network (Chapter 2 and 3). In light of this, I hypothesised that ecosystem processes would also be more variable at smaller, upstream sites. I predicted that the rates of ecosystem processes would respond directly to patterns in nutrients and in-stream conditions and that this may be demonstrated as an indirect response to land-use and landscape variables that govern the conditions in the stream at any point in the catchment. I also predicted that due to the flowing, connected nature of the system, some amount of variation in the patterns of ecosystem processes would be purely spatial, related to sites nearby. The extent of spatially and environmentally-related variation in ecosystem processes was examined for the suite of variables and for each variable individually. I predicted that some stressors would be associated with all measures, but individual measures would show unique variations in response to different impacts. The level of spatial and environmental structuring of each measure of ecosystem processes was likely to vary and chlorophyll and the DIN:SRP ratio were likely to be more spatially structured; due to the flow-through of solutes and colonising propagules.

4.2 METHODS

4.2.1 Study system and site selection

I sampled four measures of ecosystem productivity at sites in the Cam River during February and March, in the austral summer of 2010, when base flows are common (Biggs et al. 1990). Physico-chemical water quality was sampled synoptically two times in this period. In-stream, riparian and land-use variables were also measured to be used as predictors of ecosystem processes in analyses.

The Cam River arises from numerous springs (39 found by Environment Canterbury to date, Environment Canterbury GIS layers) in and around the town of Rangiora, 25 km north of Christchurch in the South Island of New Zealand (43.3034°S 172.5914°E). These springs coalesce to form the four major branches; Southbrook, Middlebrook, Northbrook and Coldstream (Figure 1). These main stems confluence in relatively quick succession to the south-east of Rangiora, forming the Cam River, which joins the Kaiapoi River and flows into the sea through the mouth of the Waimakariri River.

Riparian plant species naturally occurring in the area are flax, toetoe, grasses and sedges (Winterbourn 2008). The Cam River varies in its degree of naturalness throughout, from stoney-bottomed meandering channels to drains that are straightened and excavated. Water-races, fast flowing and of uniform depth, also enter this system. As in other lowland streams on the Canterbury Plains, parts of the Cam River have an abundance of algae and rooted macrophytes, which are home to a specific set of invertebrates and are periodically cleared by councils and farmers (Winterbourn 2008).

The Cam River and its catchment have a long history of mixed land use. The area was first settled by Europeans in 1851. At this time the river ran through native forest, wetlands and

grasslands (Hawkins 1957). Several mills profited from the water in swamps and small creeks that drained into the Cam River. These produced a variety of wastes, many of which were disposed of into the river. Agriculture and farming became the primary industries after water from the swamps was drained and soils became more productive (Hawkins 1957). In more recent years, the non-urban parts of the catchment have become dominated by dairy farming and lifestyle blocks (Biggs 1985).

All waterways flowing into the Cam River system were mapped; this included all drains, channelised stream segments and water races or irrigation canals that entered the system. A GIS layer of all water flowing into the Cam River was built by tracing the river lines using Google Earth images (the most accurate and high quality images available), in combination with field knowledge and ground truthing, using a GPS (Garmin GP560) (Figure 1). Sixty-one sites were spread throughout the Cam River system as per the "snapshot" methodology (Chapter 1) (algal production was only measured at a subset of these sites). With this method I aimed to produce an instantaneous picture of all concentrations and fluxes in the watershed, by sampling every confluence and discharge point within a short time period during base flow (Grayson et al. 1997). For ecosystem processes this allowed coverage of all parts of the system where conditions may potentially change. Sites were placed, where feasible on all first-order tributaries and point sources entering the system. They were situated at least 25 m above and below every point of confluence, so that no two sites were less than 50 m apart. Sites were also placed along uninterrupted main stem and tributary reaches, approximately 500 m apart (Figure 1).

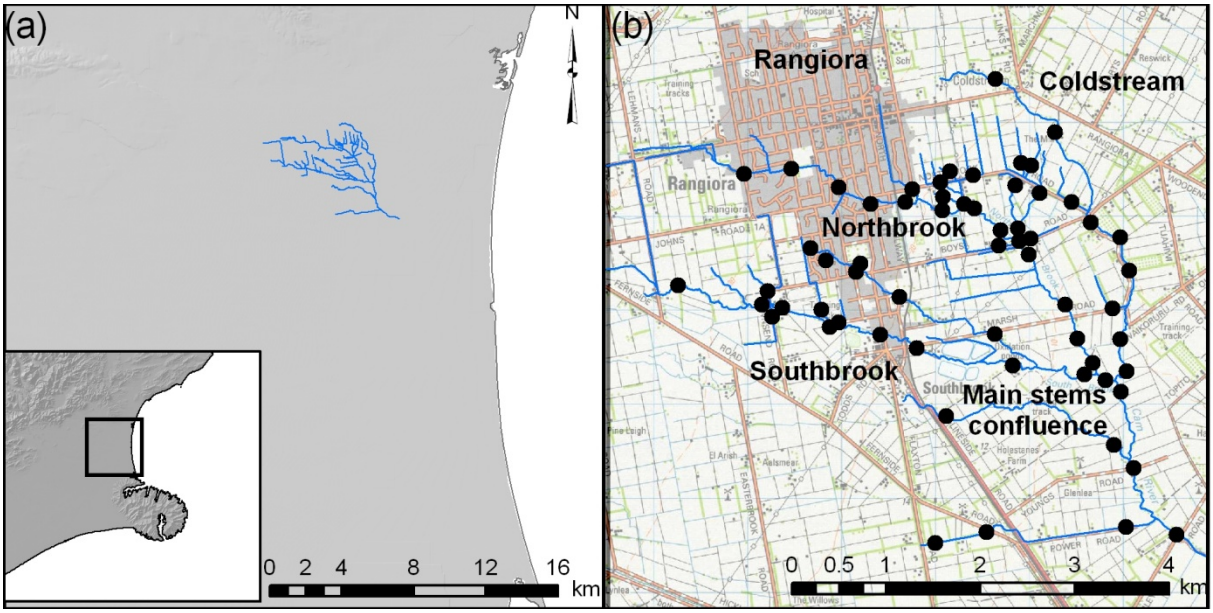


Figure 1: The location of the Cam River on the Canterbury Plains (a) and the location of the Cam River sampling sites (b). The blue lines show the natural channels of the Cam River system (constructed by tracing from Google Earth images and ground truthing). The stream drains towards the bottom right. The river lines and sampling sites are overlain on a topographical map of the area. The nearby town Rangiora and three major branches of the river are identified, these branches confluence in quick succession to the south of the town, to form the main Cam River.

4.2.2 Measuring ecosystem processes

4.2.2.1 Algal productivity

I used levels of chlorophyll *a* in caged, unglazed ceramic tiles (colonisation surfaces) as relative measures of algal growth between the sites. The tiles (5 x 5 cm^2) were positioned inside cages (250 ml plastic containers with four, 5 x 5 cm^2 mesh panels) which were glued inside polystyrene floats (rings, 10 cm inner radius, 15 cm outer radius and 3 cm deep), attached with a cord to a metal stake in the streambed. In this way the tiles at all sites remained at 10 cm depth below the surface and were able to sway in the current, deflecting debris and were not accessible to grazing invertebrates. Water velocity in the containers was reduced, but proportional to the surface current velocity. The intent of placing the tiles in the floats was to prevent insect colonisation and consumption of the

algae, so that the algal growth would primarily reflect the water conditions in the stream. At each site, three replicate floats, with 1 tile in each, were placed every 5 m, at random offsets from the stream bank. At each float, the percentage of overhead shade cover was estimated visually, water depth read on the downstream side of a ruler and water velocity measured, using a Marsh-McBirney Flow Mate, at the stake (at 4/10ths of the water depth) and the float (directly upstream of a window in the plastic container, to measure velocity across the tile). After three weeks exposure (late February to March) the tiles were removed, kept on ice in the dark and then frozen.

The chlorophyll was extracted in 70 ml of unbuffered 90 % ethanol over 24 hours at 4 °C (adapted from Biggs and Kilroy 2000). Three subsamples of 3 ml of extract were gently removed, so as not to disturb sediments, transferred to a 1-cm glass cuvette and the absorbance was read at wavelengths, 665 nm and 750 nm (turbidity correction), using a uv-visible spectrophotometer (Shimadzu, PharmaSpec uv-1700). The extract was then acidified with 0.3 ml of 0.1N HCl (to correct for the measurement of pheopigments) and absorbance re-read at the previous wavelengths (Biggs and Kilroy 2000). Chlorophyll *a* was calculated in $mg^{-1}m^2$ using the absorption coefficient for chlorophyll *a* as defined by Sartory and Grobbelaar (1984) and averaged across tile extract replicates and within each site.

4.2.2.2 Leaf decomposition and microbial respiration

I used packs of willow leaves to measure variation in the rate of leaf breakdown and microbial respiration on leaves, between sites, on the Cam River. Willow (*Salix* spp.) leaves were collected the previous autumn from the University of Canterbury campus and air dried completely. Each pack consisted of a 5 g (±0.1 g) portion of willow leaves (weighed on an analytical balance) packed inside a fine mesh onion bag (1 mm mesh) (Benfield 2006). Three packs per site were attached individually to metal stakes, driven into the streambed every 5 m, at random offsets from the stream bank (algal tile floats attached to the same stakes). Areas likely to be unstable or eroded were

avoided. The packs were attached so as to float just above the stream bottom, downstream of the stake to prevent accumulation of debris. The water depth at each pack was read on the downstream side of a ruler and water velocity measured using a Marsh-McBirney Flow Mate (at 4/10ths of the water depth). The leaf packs were collected after four weeks, and transported on ice to the laboratory. At the time of collection, 500 ml of filtered stream water was collected from each site in acid washed plastic bottles and transported on ice to the laboratory. Within 2 hours of collection, each pack was opened and 3 leaves (or partial leaves where no whole leaves remained) were removed in order to study microbial respiration rates on leaf material between sites.

Oxygen consumption was measured on the cores taken from the leaf litter as a measure of microbial activity on the decomposing litter (methods adapted from Niyogi et al. 2001). The filtered stream water was kept at 15 °C and oxygenated to the point of saturation (at least 30 minutes), using a central air system prior to use. The leaves were gently rinsed in filtered stream water then two 1 cm diameter cores were removed from each leaf with a metal corer and transferred to a 35 ml plastic vial (autoclaved previously at 120 °C for 15 minutes for sterilisation). As not all packs contained enough complete leaves to take full cores, in some instances the material consisted of several partial cores which approximated the surface area of two complete cores. Vials were then filled with saturated, filtered stream water from the corresponding site and covered tightly with parafilm, avoiding air bubbles. The incubations were performed in 15°C water-baths to prevent diffusion of oxygen into the vials through the plastic, and lasted 30-40 hours. Niyogi et al. (2001) found that oxygen uptake on willow leaves was linear during the course of incubations. Vials were gently agitated during the incubation to prevent the formation of a diffusion gradient (Niyogi et al. 2001). Additional treatments of vials containing only saturated, filtered stream water were run as controls to ensure that oxygen uptake was not significant compared with that of litter samples (Niyogi et al. 2001). At the end of the incubation period, oxygen levels in each vial were measured (YSI 550 dissolved oxygen meter) and the leaf material from each vial dried and weighed. Microbial

respiration was averaged within packs and sites and calculated as the micrograms of oxygen consumed per milligram of leaf matter per hour.

The amount of leaf litter break-down that had occurred at each site was calculated from the change in leaf mass during the treatment. Leaf litter from each pack was passed twice through a 2.5 mm sieve, washing gently, to remove small invertebrates and all forms of fine particulate matter. Macroinvertebrates, stones, sticks, leaves of different species and any other material obviously foreign to the litter pack were removed and the remaining willow leaves dried at 50 °C for at least 24 hours (Benfield 2006). As many packs contained a large amount of fine sediment, the leaves were weighed, then combusted in an ashing oven (McGregor Muffler, 550 °C for 2 hours) and reweighed to calculate the ash free dry mass (AFDM) of the leaf material. The difference in weight, before and after ashing, was used to calculate the mass of sediment that was attached to the leaves; this was divided by the AFDM to give a proportional weight of sediment accumulation.

4.2.2.3 Trophic state

Considerable additional information about the trophic status of a river can be obtained from the ratio of nitrogen to phosphorus in stream water. Spatial variation in the trophic state throughout the Cam River was studied using three snapshot samples of the water quality (Grayson et al. 1997). There is disagreement about which measures should be used to form the ratio (Grayson et al. 1997), but in this study I use the ratio of dissolved nitrate and nitrate (assumed as equivalent to DIN, due to the small proportion that was nitrite) to dissolved phosphorus (SRP), as these are the major forms of nutrients available for algal growth. It has been suggested that this is the best representation of the trophic status, as experienced by productive biota. The methods associated with the collection of the snapshot water quality data will be described in the following section. The interpretation of the trophic state of the Cam River was based on the assumption that for algal growth the ratio should be

16:1 mol of N to P. A ratio of less than 10 indicates nitrogen limitation and a ratio of greater than 20, phosphorus deficiency (Stelzer and Lamberti 2001).

4.2.3 Predictor variables

4.2.3.1 Physico-chemical water quality

Water samples were collected in two events (February and March), spread over 1 to 4 days, at the start and end of the deployment of the algal tiles and leaf packs. To meet the requirements of snapshot sampling (Chapter 1), sampling was conducted within a period of base flow, defined as the period between storms when the hydrograph was in the later stages of the recession limb, considered as at least two days after a storm flow peak (Pionke et al. 1999). Sampling was not conducted if rain had occurred in the previous week in the catchment, Southern Alps or foothills which may cause the Waimakariri and Ashley Rivers to rise, thereby raising the water table feeding the springs in the Cam River. In light of the relatively small variation in discharge across each sampling period, I considered flow in the Cam River to be stable during each sampling period. Sampling was only conducted up to 5 hours either side of midday to minimise diurnal variation in parameters such as dissolved oxygen and temperature. Diurnal variation in water chemistry is minimal with regards to all major forms of nitrogen and phosphorus except for ammonia (not analysed here) (Finlay et al. 2011). I measured eight physical and chemical properties of the water, as well as discharge, at each visit to a site (field and laboratory processes described in Table 1). Each day I visited downstream sites first and collected water samples immediately on arrival at a site to avoid contamination from sediments disturbed by entering the water. The average values of the sampling events were used when analysing, as this reflected the overall conditions each measure was exposed to.

4.2.3.2 Other in-stream variables

Other in-stream and site properties, which were considered more permanent were assessed on only one occasion (methodology in Table 1). Macrophyte cover, algal cover and the proportion of the site shaded were estimated visually (Table 1). Stream size-related variables include stream bank full width, stream order and discharge. Although discharge was measured during each visit to a site, average discharge was used in analyses as this better described the conditions during each measure of ecosystem processes. Water temperature loggers (HOBO pendant temperature/light, Onset Corporation) were installed underwater, just above the streambed at a subset of sites, covering the period of algal tile and leaf pack deployment. Data was logged every 30 minutes. Data from the three week period when the algal tiles and leaf packs were in the stream was averaged (5 hours either side of midday) for consistency with spot measurements. As average spot temperatures during the site visits correlated well with average temperatures from the loggers, I estimated the temperature at sites without loggers by averaging the spot temperatures.

Table 1: Field and laboratory methodology for the eight water quality variables and other predictor variable measured at each site, as well as treatment in analyses.

Variable	Units	Field and/or laboratory methodology	Reference	Treatment
Temperature	ºC	Time averaged spot measurement. The YSI Sonde 6600, a multi-parameter, water quality measurement device was placed in the thalweg of a run at the top of each site, continuously logging during time spent at each site. The final 5-20 minutes of data from each site was averaged (visit times varied depending on sample run intensity).		Eight response water quality variables
Dissolved Oxygen	mg/L			
pH				
Conductivity	µS			
Turbidity	NTU			
Dissolved Nitrite-N + Nitrate-N (DIN)	mg/L	I collected 80 mL of water in a syringe from the thalweg of the stream just below the surface and pushed through a filter (Whatman GFF 250 Millipore rating) into a 100mL opaque plastic bottle (pre-soaked in 5% hydrochloric acid overnight then rinsed three times with distilled water and a further three times with milli-q water). 80 mL of unfiltered water was collected in a similar way. Samples were kept on ice and then frozen. Filtered samples were thawed then analysed colourmetrically for DIN (assumed from combined nitrate-nitrite due to low nitrate concentrations). SRP using an automated, high throughput, water chemistry machine, the Easychem Plus (Systea, Italy). All chemicals used to make standards and reagents were of reagent grade and methods used by machine standard. The unfiltered water samples were thawed then processed for total phosphorus (TP) using standard colourmetric methods and testing for absorbance on a Trilogy Laboratory Fluorometer using a PO4 module.	American Public Health Association (APHA), 1995	
Soluble reactive phosphorus (SRP)	µg/L			
Total phosphorus (TP)	µg/L		Wetzel et al. 2000	
Algal cover*	%	Visual estimate of % of the stream bed covered by algae.		Predictor variables
Macrophyte cover*	%	Visual estimation of % of the stream bed covered by macrophytes.	Harding et al. 2009	
Shading*	%	Visual estimate of % of the stream bed shaded when the sun is overhead.		
Discharge	m^3/s	Discharge measured across one run, evenly flowing and free of obstructions. Offsets were placed wherever depth or discharge changed noticeably, with no fewer than five per transect. Water depth read on the downstream side of a ruler and water velocity is measured four-tenths of water depth up from the bed using a Marsh-McBirney Flow Mate. Discharge was calculated based on standard methods.	Gordon et al. 2004, Harding et al. 2009	
Wetted Width	m	Wetted width was measured at five locations along the 20m reach.	Harding et al. 2009	

* Variables considered constant, only assessed on the March sampling run.

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4.2.3.3 Land use, riparian conditions and other GIS derived variables

For each site, catchment area was defined using the River Environments Classification (REC), a GIS-derived database of river and stream networks in New Zealand (Snelder et al. 2005). Due to the low topography of the Plains, catchments generated from digital elevations only approximate actual catchment areas, so manual changes were made to fit known water movement established through ground truthing. This process was further complicated by the stock water-races and irrigation canals, which resulted in considerable cross-catchment transfer of water. To estimate the relative inputs of these artificial waterways I used an upstream distance measure which included the length of the water channels races (not sampled) that entered the system. All parameters related to stream distance were based on the stream map constructed from Google Earth images (Google Inc. 2009) and field knowledge. All parameters relating to Euclidean distance are based on the Cartesian Co-ordinates of the sites. For each site, measures of land use, riparian conditions, road density and in-stream conditions at upstream sites were derived for the catchment area contributing to the site and the area upstream area 100 m either side of the stream, using ArcMap 9.2 tools and the New Zealand Land Cover Database ver.2 (Terralink 2004 – Table 2).

Table 2: Distance, land use buffer zone and other site-wise variables derived by GIS for use as predictor in analyses, where no units are given the variable is a proportion.

Site-wise Variables	Methodology/calculation (all variables are based on the stream maps I constructed)	Units
Stream distance to most downstream site	The length along the stream channel to the most downstream site (Bramleys Rd)	km
Upstream natural channel length	The length of all natural upstream channels (including all tributaries) based on traced and ground truthed stream maps. The decision was made to "cut off" a natural stream reach (beginning the "unnatural" portion), when upstream reaches took the form of highly channelised, free-flowing canals, of uniform depth, characteristic of water-race or irrigation networks (Google Inc. 2009, ArcMap 9.2).	km
Upstream channel length including water race	The length of stream including all natural and man-made waterways upstream of the site. Where branches split in two in the direction of flow, I only measured the branch connecting to the system. (Waimakariri District Council water race data in geospatial form and ground truthing).	km
Upstream junctions	Counted manually (excluding the water race network).	
Stream order	Counted manually (excluding the water race network).	
Catchment area upstream	Catchment areas were defined using the River Environments Classification (REC), then altered manually to meet known directions of water flow and channels not delineated by this model (Snelder et al. 2005).	km^2
Area of 100-m buffer zone upstream	The area upstream of each site within 100m of the natural stream. This distance was chosen based on its use in buffer zone delineation (Baker et al. 2007) and as the smallest distance class to measure land use metrics to assess critical distances of impact (Houlahan and Findlay, 2004) (ArcMap 9.2).	km^2
Length of road per km^2 of upstream catchment area	The total length of all paved and meteled roads in each upstream catchment area Zealand Landcover Database ver.2, (Terralink 2004), expressed as a fraction of the area in km^2.	
Proportion upstream catchment area that is urban	The proportion of the catchment area that is built-up (New Zealand Landcover Database ver.2, Terralink 2004))	
Proportion upstream buffer zone that is urban	As above but for the 100-m buffer zone.	
Proportion upstream catchment that is in moderate to high intensity farming	The proportion catchment area that is in use as Dairy, Beef, Sheep or Sheep and Beef (Agribase 2009)	
Proportion upstream buffer zone that is in moderate to high intensity farming	As above but for the 100-m buffer zone.	
Proportion upstream in each of each of 6 land use types	The proportion catchment area that is in use as dairy, sheep, beef, sheep and beef, arable and lifestyle as defined by the AGRIBASE (Agribase, 2009)	
Proportion upstream buffer zone in each of each of 6 land use types	As above but for the 100 m near zone.	
Proportion upstream catchment with forest cover	The proportion of the catchment covered in exotic or native forest (New Zealand Landcover Database ver.2, (Terralink 2004))	
Proportion upstream buffer zone with forest cover	As above but for the 100-m buffer zone.	
Elevation change within catchment.	Based on the reach of highest and lowest elevation in all mapped stream reaches that correspond to REC reaches (Snelder et al. 2005).	m
Average level of macrophyte cover of a upstream sites	An average of the percentage cover of macrophytes of all upstream sites (Table 1)	%
Average level of shading cover of a upstream sites	An average of the percentage cover of shading of all upstream sites (Table 1)	%

4.2.4 Analyses

4.2.4.1 Patterns in ecosystem processes

To initially evaluate system-wide patterns in ecosystem processes, each measure of ecosystem processes was expressed visually using graduated colour symbols to represent the value of each site, at its actual location on a map of the streamline (as in Tu and Xia 2008, ArcMap 9.2). Correlation between the ecosystem process measures was assessed using the non-parametric, Spearman's rank correlation procedure (two-tailed, $\alpha = 0.05$), to overcome problems associated with lack of normality.

To test whether variability in each ecosystem process changed with stream size, I fitted linear functions to the upper and lower limits of relationships between each ecosystem process and discharge using quantile regression in R ('quantreg' package; Cade and Noon 2003, R Development Core Team 2007, Koenker 2009). Discharge, width, stream order and kilometres of stream channel upstream of a site aligned along the same axis under principal components analysis, so discharge was a suitable measure to generally represent stream size. I also tested if discharge was related to these limits by investigating limit responses for each measure of ecosystem processes when related to stream size, by regressing the sites that fell within upper (90^{th}) or lower (10^{th}) quantiles (depending on which relationship was investigated) of the ecosystem process-size relationship against discharge. Confidence intervals were computed by the rank inversion method and P-values by bootstrapping (Koenker 2009).

4.2.4.2 Partitioning the variation to space and environment

One way to estimate the relative contribution of the environment (water quality, land-use and in-stream conditions) and other spatial processes to patterns in ecosystem processes, is to partition the variation measured between environmental and spatial factors, using spatial eigenfunctions (Cottenie 2005, Peres-Neto and Legendre 2010). This method is related to the method used in community ecology to

separate the contribution of niche processes (environmental conditions) and other spatial processes (such as dispersal) on community structure (Borcard et al. 1992, Legendre et al. 2005). Provided that the environmental factors have been appropriately quantified (with relevant variables), the extent of their influence can be quantified, and residual spatial variation attributed to other spatial processes, such as flow of solutes and matter within the system. I have used this method to partition the variation in ecosystem processes into four fractions; pure environmental (E|S), pure spatial (S|E), jointly explained (environment-spatial) and unexplained variation (Legendre et al. 2005). It should be noted that part of the environment-space fraction could be due to other spatial processes that show correlation with the environment (Bell et al. 2006) and that the pure spatial fraction may hide the effects of some unmeasured spatially structured environmental variables (Borcard and Legendre 1994, Jones et al. 2008).

I used Eigenfunction analysis to represent the spatial fraction. This involved constructing a series of spatial eigenvectors using principal coordinates of neighbour matrices (PCNM) and related methodologies (Borcard and Legendre 2002, Dray et al. 2006). These methods are based on, and comparable, to the Moran's I statistics, which are the most commonly used statistics for spatial autocorrelation analysis (Rangel et al. 2006). The PCNM method codes spatial information in a way that allows one to recover various structures over the whole range of scales that the sampling design passes (Borcard and Legendre 2002). Empirical results show that the positive eigenvectors alone give a good representation of the spatial relationships (Borcard and Legendre 2002, Borcard et al 2004). These eigenvectors represent an orthogonal spatial structure (as they are the product of a symmetric matrix) and can be used as explanatory variables in regression or canonical models (Dray et al. 2006). Eigenvectors (constructed by any method) with large eigenvalues describe global structures, whereas those with small eigenvalues describe local structures (Borcard and Legendre 2002).

I used eigenfunction-based techniques to create a series of metrics to describe the spatial structure in ecosystem processes, based on different distance measures. I then assessed how well each metric accounted for variance in the ecosystem processes as a multivariate suite and each individually (Griffith and Peres-Neto 2006, Blanchet et al. 2008b). I used both Euclidean distance (PCNM-E) and stream channel distance (PCNM-S), to construct two sets of spatial eigenvectors using the basic PCNM methods, to test the importance of overland, direct distances and in-stream distances, respectively, on structuring ecosystem processes, considering all sites as connected (Table 3).

This basic framework has been developed to include options for variably connected and weighted spatial representations, with the potential to accurately represent processes in stream networks (Dray et al. 2006). Another set of spatial eigenvectors were created with the Moran's eigenvector map (MEM) method using a binary connectivity matrix and a weighting matrix (stream distance between sites) (Dray et al. 2006). This allowed variable connectivity and rate of transference between sites to be taken into account in the construction of this metric (MEM, Table 3). A weighted metric was constructed by multiplying a vector of weights to the table of connectance used in the MEM (Dray et al. 2006). I used average discharge to weight the links between sites to more accurately represent the rate at which solutes and particles in the water are transferred throughout the system (MEM-W, Table 3).

Table 3. Four different spatial metrics were tested on sites in the Cam River. Eigenfunction-based spatial filtering techniques were used, allowing flexibility of weighting and connectivity in spatial representation. I used principal coordinates of neighbour matrices (PCNM) with two distance metrics, and stream distance for the other spatial metrics. I used symmetrical distance between flow connected sites, based on Moran's eigenvector maps (MEM) and a corresponding weighted metric (MEM-W, based on the average velocity encountered between sites) to represent different spatial structuring processes.

Distance measurement	Weight	Code	Description
Euclidean distance		PCNM-E	PCNM on Euclidean distances between sites
Stream distance		PCNM-S	PCNM on stream distances between sites
Direction – flow connectance		MEM	MEM all sites directly connected by flow in both directions
	Average discharge	MEM-weighted	MEM – weighted by average discharge encountered between sites.

The spatial metric that best described variation in ecosystem processes was chosen using the adjusted coefficient of multiple determination (R^2_a) to compare the variance explained by each spatial metric (Peres-Neto et al. 2006, Blanchet et al. 2008a). Each set of eigenvectors from each of the eigenfunction analyses for each distance measure, was subjected to forward selection ($\alpha < 0.1$) to detect eigenvectors explaining the most variance in community structure (Blanchet et al. 2008a). The suite of ecosystem processes, then each individually, were analysed as functions of the set of PCNM, MEM and AEM eigenvectors by canonical redundancy analysis (RDA), a multivariate regression based analysis using the spatial vectors as predictors for the ecosystem processes. The forward selected spatial variables that explained the most variation were further used in variance partitioning.

I assessed the relative contribution of environmental processes versus other spatial processes on multi-scale patterns in ecosystem processes using partial redundancy analysis (pRDA (Cottenie 2005, Dray et al. 2006, Peres-Neto and Legendre 2010). The set of environmental variables used in each analysis was chosen by forward selection in relation to the response variable(s) in question (as for the spatial variables, Blanchet et al. 2008a). I attempted to isolate the spatial and environmental contributions to the rates of ecosystem processes by including not only the environmental variables that impacted each site locally (Table 1), but also a range of variables which defined the condition of the

catchment feeding water to that site, and that of the riparian zone along upstream reaches (Table 2). Sets of three matrices were used in pRDA, a spatial matrix (forward selected spatial variables from the best spatial model) and environmental matrix and a response matrix (ecosystem processes or a single variable for individual tests), to compare proportions of variation in the community explained by each predictor variable matrix. The relative weight of each fraction of partitioned variance was represented by the adjusted R^2 (R^2_a) which controls for the number of sites (differing for chlorophyll), the number of explanatory variables and the probability of detecting effects (Peres-Neto et al. 2006). The forward selected environmental and spatial variables were first tested as non-saturated global models, individually ([S] and [E] fractions). For each analysis, if either the global spatial model or global environmental model was significant, I proceeded with the partitioning approach to test the relative proportions of their independent effects on community structure (independent spatial [S|E] and independent environmental [E|S] fractions) and finally the joint variation shared by space and environment (Laliberté et al. 2008).

These analyses were conducted in the R-language environment (R Development Core Team 2007) using the packages "vegan" (Oksanen et al. 2007) for RDA and variation partitioning; "PCMN" (Dray et al. 2006) for the construction of PCNM variables; and "packfor" for the selection of explanatory variables in the RDA. In all tests of significance, 999 permutations were used. Following Anderson and Legendre (1999) permutation of raw data is adequate for ANOVA as there are no outlier values in the factors.

4.3 RESULTS

4.3.1 Patterns in ecosystem processes

The abundance of chlorophyll *a* was significantly positively correlated with the reduction in organic matter in leaf packs ($r_s = 0.369$, $p = 0.009$), but no significant correlations occurred between the other measures of ecosystem processes. Algal productivity was highest at sites in and around Rangiora, and in main stems directly downstream of the town (Figure 2a); correspondingly the reduction in leaf mass was also higher at main-stem sites downstream of Rangiora, compared with those above the town and in the Coldstream branch (Figure 2b). Microbial respiration and the DIN:SRP ratio became lower downstream, with various patches of lower levels punctuating the generally high levels in the upper system (Figure 2c and d).

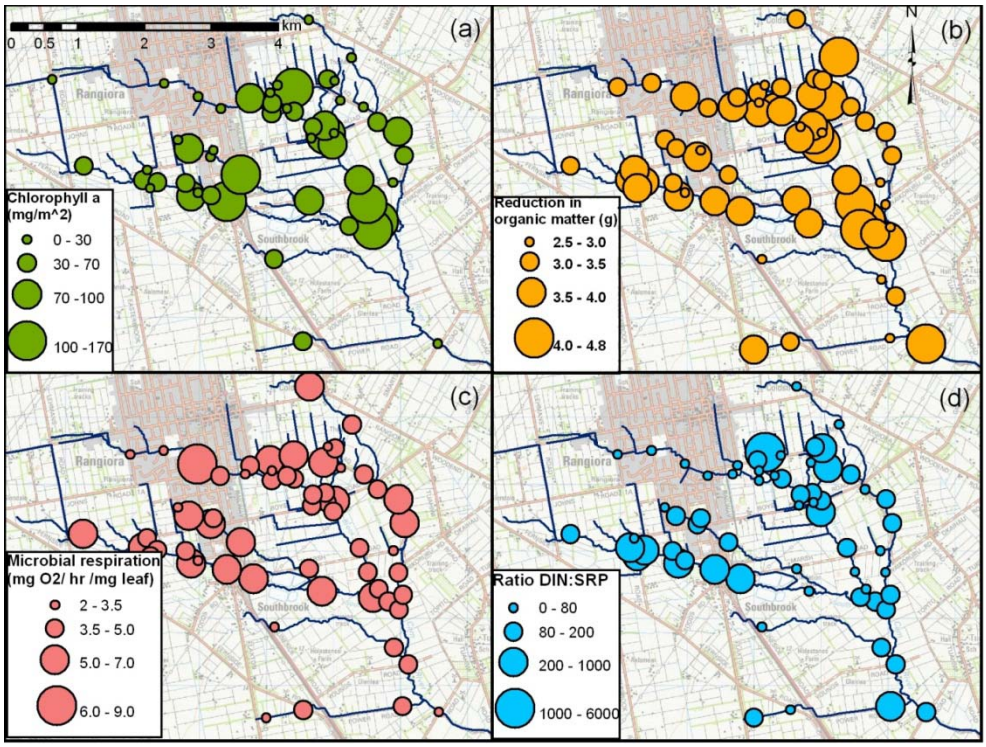


Figure 2 – Maps showing the spatial variation in each measure of ecosystem processes; chlorophyll *a* concentration (algal productivity) (a), reduction in organic mass of leaf packs (b), the rate of microbial respiration on those leaves (c) and the DIN:SRP ratio (d), throughout the Cam River. The stream drains to the bottom right. The size of each circle represents the magnitude of that parameter at that site (see individual keys for actual levels), the divisions were created using the "Jenks" natural breaks function in ArcMap 9.2. The scale bar and North arrow apply to all maps.

There was a trend in all four measures of ecosystem processes towards less variation as average discharge increased (Figure 3a-d). The relationship between organic matter loss in leaf packs and discharge had a significant positive floor and ceiling, although only the floor is shown, as this is where the limit relationship appears most observable ($p < 0.001$, Figure 3a, Table 4). The relationship between chlorophyll *a* and average discharge also had a significant positive floor limit response ($p = 0.015$, Figure 3d, Table 4). Alternatively, the relationships of DIN:SRP and microbial respiration with discharge has a negatively sloped upper ceiling (c and b). The response of these ecosystem processes to stream size measures was only weakly significant ($\alpha < 0.1$).

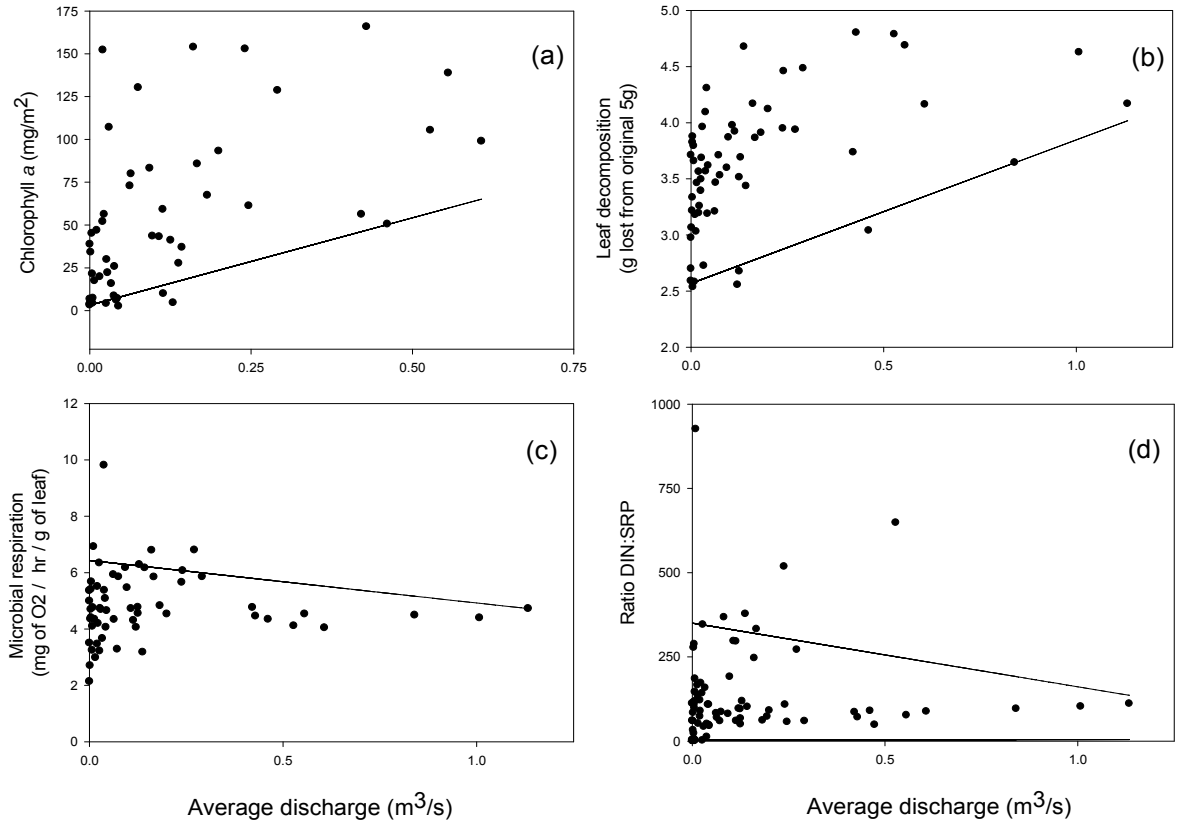


Figure 3 – Quantile regression relationships between average discharge and algal productivity (as chlorophyll *a*) (a), leaf decomposition (reduction in organic matter in leaf packs) (b), microbial respiration on this leaf matter (c) and the DIN:SRP ratio (d). The lower quantiles (10^{th}) in (a) and (b) were significant at $\alpha < 0.05$ and the upper quantiles (90^{th}) in (b) and (c) were significant at ($\alpha < 0.1$). Note that Chlorophyll was measured at a subset of sites, thus the discharge axis is shorter.

Table 4 - Results of quantile regression, where discharge was the predictor variable for each response ecosystem process variable. Values highlighted in bold were significant ($\alpha < 0.05$) when assessed using bootstrapping methods.

Response	Quantile	Intercept	Slope
Chlorophyll	Upper 90^{th}	**100.41**	217.41
	Lower 10^{th}	3.22	**102.25**
Microbial respiration	Upper 90^{th}	**6.43**	-1.51
	Lower 10^{th}	**3.49**	0.21
Leaf decomposition	Upper 90^{th}	**3.90**	**1.68**
	Lower 10^{th}	**2.57**	**1.28**
Ratio DIN : SRP	Upper 90^{th}	**76.18**	350.64
	Lower 10^{th}	**1.97**	-189.27

4.3.2 Partitioning the variation in ecosystem processes

The Moran's Eigenvector Map (MEM) method consistently produced spatial vectors that described the largest amount of variation in ecosystem processes, when compared to the other metrics (except the multivariate analysis, where the discharge-weighted MEM method produced the best spatial metrics) (Table 5). A similar pattern in values of significant eigenvectors chosen by the forward selection procedure occurred across all response variables. Two small blocks of low value eigenvectors were significant, describing local spatial structures and several blocks from approximately 0.5 to 50 - 100, describing intermediate to global spatial structures (Table 5). The significant spatial eigenvectors associated with the multivariate ecosystem process had eigenvalues that covered the widest range (0.002 – 623). These values were also spread more evenly across all spatial scales than those of the other response variables (Table 5). The spatial vectors describing the variation in chlorophyll *a* had values covering the smallest range, with a smaller block of values describing the global structures than the other variables, and a slightly lower upper limit of values than the other variables (the upper limit of the DIN:SRP ratio is also slightly lower) (Table 5). Approximately twice as many environmental variables were selected as contributing significantly to the variation in leaf decomposition and the DIN:SRP ratio compared with the other ecosystem processes (Table 5). The environmental variables of stream order and DIN were chosen recurrently as explaining a significant amount of the spatial variation in ecosystem processes (for 3 out of the 5 response variables), as were other variables relating to stream size, macrophyte cover and catchment land-use as sheep farming (Table 5). The relationship between these environmental variables and each measure of ecosystem processes varied (Coefficient, Table 5). Although some variables were chosen more frequently, a unique set were chosen by the forward selection process to describe each measure of ecosystem processes.

The variance in ecosystem processes explained by spatial structure in all global models was highly significant ($p \leq 0.01$, Table 6). Environmental factors were also significant in explaining variation in

ecosystem processes in all global models, but less variance was explained ($p \leq 0.05$, except for DIN:SRP ratio at $p \leq 0.1$, Table 6). All response variables, except leaf decomposition, had more variance explained by the independent effects of spatial structure (S|E) than the independent effects of environmental variables. Thus the portion of variance explained by S|E was highly significant in all instances ($p \leq 0.01$, Table 6, Figure 4). In the case of chlorophyll *a* and the DIN:SRP ratio, the variance explained by spatial structure was much higher than explained by environmental factors, evident in the high ratios of S|E and E|S (Table 6). Although more variance in the water quality was explained by spatial structure, there were significant independent environmental effects that simultaneously explained portions of the variance for leaf decomposition ($p \leq 0.05$, Table 6, Figure 4c), the combined ecosystem processes and microbial respiration (although only weakly so $p \leq 0.1$, Table 6, Figure 4a and d). Variation jointly explained by environmental and spatial predictors, which could not be statistically separated into independent effects of either space or environment (Peres-Neto and Legendre 2010), was relatively high. This was especially true for leaf decomposition, where joint variation exceeded both S|E and E|S portions of variation (Joint, Table 6). The variation jointly explained exceeded that explained by the independent effects of environment in all cases (Table 6). This meant that spatial processes were generally more important in determining ecosystem processes than local environmental conditions

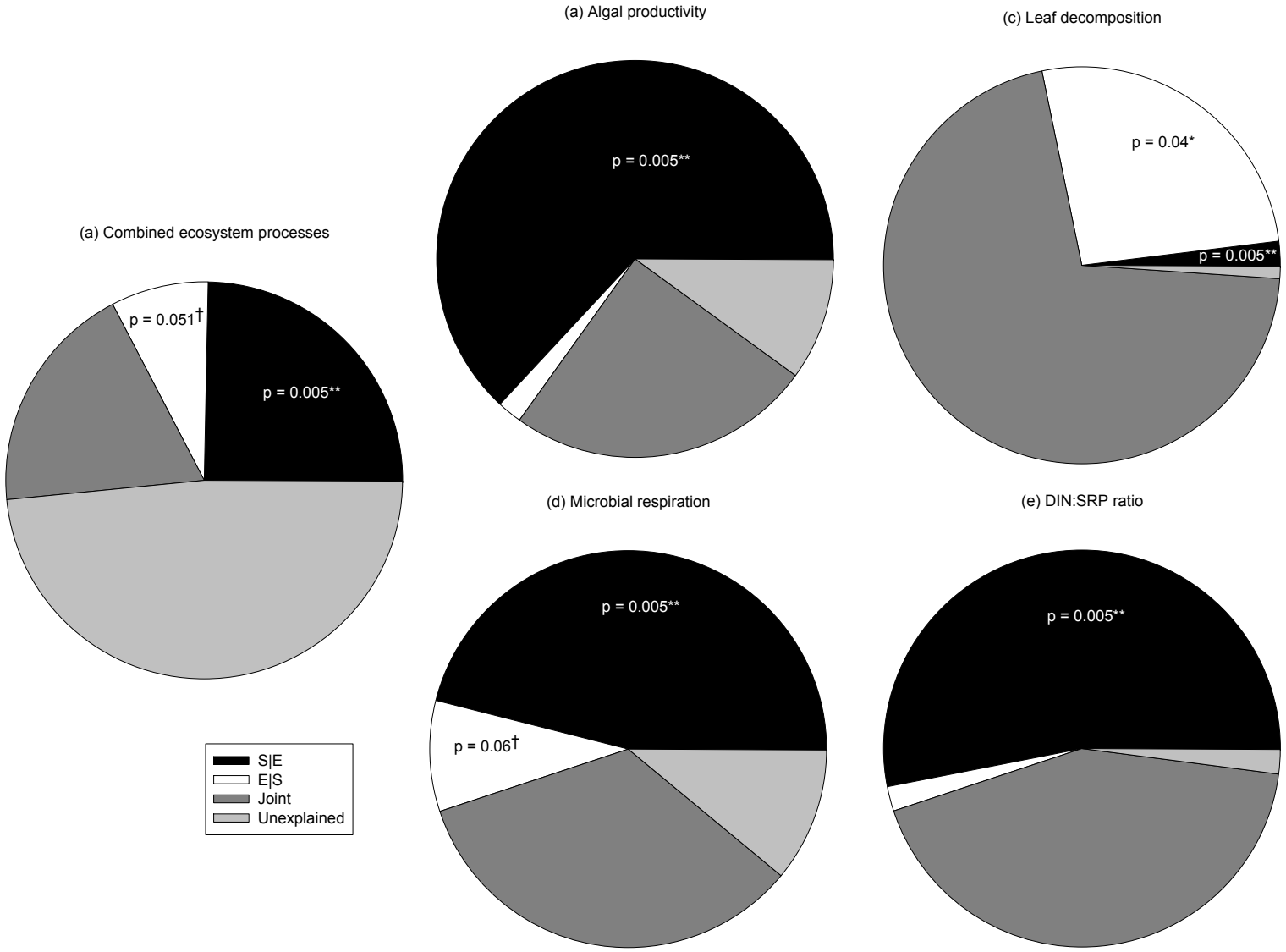
Table 5. Spatial and environmental variables used in the variance partitioning of the combined ecosystem processes, analysed together and in analysis of each ecosystem process variable individually. The best spatial metric described the most variation in the ecosystem process(es) indicated by the highest adjusted coefficient of multiple determination (R^2_a). Refer to Table 3 for spatial metric acronyms. The "# sig. spatial" columns indicate the number of vectors that were significant during forward selection of spatial and environmental variables ($\alpha < 0.1$ and 999 permutations). The eigenvalues of the significant spatial vectors are displayed to show patterns in distribution. Eigenvectors (constructed by any method) with large eigenvalues describe global structures whereas those with small eigenvalues describe local structures (Borcard and Legendre 2002). Environmental variables that are significant in each model are listed. Note, there are no absolute units for the eigenvalues of the significant spatial eigenvectors

Response variable(s)	Best spatial metric	# sig. spatial	Ranges of sig. spatial vectors (eigenvalues on Log scale) 0.001 0.01 0.1 1 10 100	No. sig. enviro.	Significant environmental variables	
					Abbreviated name	**Coefficient**
Combined ecosystem processes	MEM-W	29		5	DIN	-0.78
					Stream order	-0.24
					Prop sheep-beef 100-m	0.18
					Conductivity	-0.13
					Site macro cover	-0.09
Chlorophyll a	MEM	24		4	Stream order	0.76
					Shading site	-0.51
					Prop lifestyle 100-m	0.30
					Prop sheep 100-m	0.04
Leaf decomposition	MEM	37		9	Stream order	0.62
					Prop vegetation area	-0.56
					Upstream km natural	0.5
					Metalled road length	0.21
					Prop lifestyle 100-m	0.21
					Turbidity	0.2
					Prop sheep 100-m	-0.13
					Prop vegetation 100-m	-0.079
					pH	-0.026
Microbial respiration	MEM	38		4	Prop sheep-beef area	-0.70
					DIN	0.47
					Prop arable area	0.36
					Shade upstream	0.33
DIN:SRP ratio	MEM	43		8	DIN	0.80
					Prop sheep area	0.45
					Conductivity	0.41
					Site macro cover	-0.14
					Prop arable area	0.13
					Prop lifestyle 100-m	0.13
					Dairy area	-0.035
					Vegetation 100-m	-0.03

Table 6. Spatial (S), environmental (E), independent spatial (S|E) and independent environmental (E|S) partitions of variation, and joint variation in ecosystem processes throughout the Cam River compared using partial redundancy analysis (pRDA). R^2_a, the adjusted correlation coefficient, was used to partition the variance into four fractions (S|E, E|S, joint and residual) in an unsaturated model (see methods text for details). Values significant at † $p \leq 0.1$, * $p \leq 0.05$, ** $p \leq 0.01$. 'Joint' variation is the component shared by spatial and environmental variation which cannot be statistically separated.

Response variable(s)	No. sites	S		E		S|E		E|S		Joint	Residual	S|E : E|S
		R^2_a	p	R^2_a	p	R^2_a	p	R^2_a	p	R^2_a		
Combined ecosystem processes	48	0.89	0.005**	0.37	0.01*	0.25	0.005**	0.08	0.051†	0.19	0.49	3.13
Chlorophyll a	48	0.88	0.005**	0.39	0.005**	0.63	0.005**	0.02	0.45	0.25	0.10	31.50
Leaf decomposition	61	0.94	0.005**	0.65	0.005**	0.02	0.005**	0.26	0.04*	0.70	0.01	0.08
Microbial respiration	61	0.97	0.005**	0.17	0.015*	0.46	0.005**	0.09	0.06†	0.34	0.11	5.11
DIN:SRP ratio	61	0.97	0.005**	0.63	0.053†	0.53	0.005**	0.02	0.16†	0.43	0.02	26.50

Figure 4: The relative contribution of the independent spatial (S|E), independent environmental (E|S) partitions of variation, joint variation and unexplained (residual) variation in ecosystem processes; as a combined measure (a), algal productivity (b), leaf decomposition (c), microbial respiration (d) and the DIN:SRP ratio (e), throughout the Cam River. This was assessed using partial redundancy analysis (pRDA). The adjusted correlation coefficient, R^2_a, was to calculate the weight of each fraction (size of the pie slice, see methods text for further details). The significance of each fraction, in each canonical pRDA model is indicated by the p value superimposed on the corresponding pie segment; $^{\dagger} \leq 0.1$, * $P \leq 0.05$, ** $P \leq 0.01$. The E|S fractions of (b) and (e) were not significant (p values not shown). Note that the joint and unexplained fractions cannot be tested for significance.


(a) Algal productivity
p = 0.005**
(c) Leaf decomposition
p = 0.04*
p = 0.005**
(a) Combined ecosystem processes
p = 0.051†
p = 0.005**
(d) Microbial respiration
p = 0.005**
p = 0.06†
(e) DIN:SRP ratio
p = 0.005**
S|E
E|S
Joint
Unexplained

4.4 DISCUSSION

The spatial pattern of ecosystem processes varied in my study depending on which component was under observation and in a way that was influenced by stream size. All spatial patterns resulted in increased heterogeneity of ecosystem processes in headwaters compared with mainstems. The directional connectivity and spatial separation of sites on the stream network interacted with environmental factors resulting in variable ecosystem processes throughout the Cam River. I used recently developed spatially explicit variance partitioning methods (Dray et al. 2006, Peres-Neto and Legendre 2010) to tease apart the relative importance of the spatial and environmental structure of the Cam River in determining the spatial patterns of each ecosystem process.

4.4.1 Patterns in ecosystem processes in the Cam River

Although the rates of algal productivity, leaf decomposition and microbial respiration on leaves in streams often have similar responses to variation in nutrient levels and temperature (Biggs and Kilroy 2004, Young et al. 2008), the only measures positively correlated with each other were algal productivity and leaf matter decomposition. This indicates that these two processes may be responding to similar stressors at similar scales. A correlation was also expected between the DIN:SRP ratio and each of the these ecosystem process measured, due to a strong link between this ratio and nutrient levels. However, the lack of a correlation in the Cam River is not surprising considering the inconsistent relationship between algal productivity (Biggs and Kilroy 2004), leaf decomposition (Young et al. 2008) and nutrients, across described studies. The lack of correlation between leaf decomposition and microbial respiration is unusual however, considering the key role the microbes often have in organic matter breakdown in streams, as well as the findings of previous studies (Young et al. 2008).

The high levels of algal productivity and leaf matter decomposition at sites in Rangiora and along the mainstems of the North and Southbrooks are in keeping with the positive correlation relationship I found between these measures. Lower levels of algal production occurred in upper Coldstream branch where shading was considerable, and in the upper reaches of the North and Southbrooks above Rangiora where flow was largely from water-races that connect with the system at these points. The amount of chlorophyll *a* measured at almost all sites in the Cam River network falls within the range associated with very impacted or eutrophic streams (25-260 $mg^{-1}m^{-2}$, mean < 70 $mg^{-1}m^{-2}$, Dodds et al. 1998). The method of assessing chlorophyll *a* I used was different from the sampling of benthic stones suggested by Dodds et al. (1998), which would have some level of grazing by invertebrates. Although my methods are not directly comparable to these guidelines, they are likely to be a better reflection of eutrophication, as they are independent of variation in grazing that may be site-specific. As the amount of chlorophyll I measured was often close to or exceeding 70 $mg^{-1}m^{-2}$, these were clearly extremely productive streams with very high algal abundance.

Although high levels of algal production are often used as an indicator of pollution in streams (Fellows et al. 2006), benthic algae provide the main source of energy for higher trophic levels in many unshaded temperate streams and their uptake of nutrients can help purify the streams (Biggs 1996). In this respect, invertebrates inhabiting reaches of the Cam River with high algal concentrations may benefit from this abundant resource. However, in parts of the Cam River, algae were so abundant that thick mats had formed, clogging the stream and very likely being difficult for the majority of invertebrates to graze because of their filamentous nature. The thick algal mats meant that many sites were heavily silted and likely had poorly functioning biotic communities as a result (Lowe and Laliberté 2008). Previous studies have found algal primary production rates to vary between the major different habitats of streams at the reach scale and that this had an impact on community study in each habitat (Keithan and Lowe 1985). Thus I would expect the large scale

differences in algal productivity in the Cam River may have flow-on effects to the distribution of aquatic organisms.

The rate of leaf litter decomposition at all sites on the Cam River was above the range indicative of good ecosystem health (0.01 to 0.03$g^{-1}d^{-1}$ Gessner and Chauvet 2002). Based on these guidelines the entire Cam River network has mild or severe impairment of ecosystem health. The rate of organic matter decomposition throughout the Cam River was much higher than necessary to maintain a healthy ecosystem (Gessner and Chauvet 2002). Terrestrial inputs, such as leaves, are an important basal food resource supporting higher trophic levels (Benfield 2006). The rapid decomposition process may mean that soluble inorganic forms of nutrients are being released into the stream, rather than incorporated into the food web as organic form.

The high level of algal production and leaf litter decomposition I observed in the mainstems of the Cam River is also shown in the positively sloped lower limit relationship with discharge, and the forward selection of stream order, (as one of the group of environmental variables) which explained a significant amount of spatial variation in these ecosystem processes in the global models and when using partial redundancy analysis incorporating spatial structure (for decomposition only). In these models, stream order was also positively correlated with algal production. These relationships imply that larger streams are likely to have higher algal cover than many small streams. This may be because larger reaches in the Cam River experience a greater flux of total and dissolved phosphorus as well as dissolved nitrate (Chapter 3) and are less shaded than smaller streams; all of which would stimulate cell growth and colony size.

The rate of microbial respiration on leaves and the DIN:SRP ratio became lower downstream through the Cam River network. In contrast to algal productivity and leaf decomposition rate, these two functional metrics displayed a negative upper limit relationship with discharge. This results in the lack of correlation between leaf decomposition and microbial respiration and indicates that sites on large reaches are less likely to have high rates of microbial

decomposition. The mismatch between patterns in microbial respiration and leaf litter decomposition suggests microbial respiration is not primarily responsible for leaf litter decomposition. Other factors, such as the density of shredding invertebrates (Baldy et al. 1995, Jonsson et al. 2001) and high flow velocity (Casas 1996) also increase breakdown rates and these factors might have a greater influence on leaf litter breakdown than the action of bacteria and fungi in my streams. The processes of microbial respiration and leaf decomposition are both likely to respond to nutrient levels however, and in the Cam River it appears that each responds in a different way.

The DIN:SRP ratio was much higher than the optimal ratio suggested as indicating healthy amounts of algal production (16N: 1P, Redfield 1958), at almost all sites in the Cam River. A high DIN:SRP ratio indicates that this system is largely phosphorus limited. However, while nutrient ratios suggest potential phosphorus limitation, absolute nutrient concentrations must also be considered. At levels above a certain threshold, limitation no longer occurs (Bothwell 1985). This may be the case in the Cam River which has relatively high levels of both DIN and SRP (Chapter 1), and may explain why algal productivity did not correlate with DIN:SRP ratio, and also why no nutrients were selected as significant in predicting variation in algal production under forward selection in my global models.

I found that variability in all measures of ecosystem functioning was higher in smaller streams than in large streams. This is likely due to the higher spatial and temporal variability of water quality in small streams (Chapter 2 and 3). The slope of the Quantile regression lines that defined the limit relationships with stream size and microbial decomposition or DIN:SRP was less steep than those of algal productivity and the rate of leaf decomposition, suggesting that stream size had less influence over the former variables. Modification of riparian zones, be it through agricultural or urban influences, results in increased stream temperature and nutrient levels, both of which stimulate productivity (Young et al. 2008). However, riparian modification can also cause

sedimentation (Gulis and Suberkropp 2003) and low concentrations of dissolved oxygen (Pascoal and Cássio 2004), which can have negative impacts on algal and microbial activity. The extremes of each physico-chemical parameter are experienced at smaller sites on the Cam River (Chapter 1 and 2). The variable response of productivity to these extremes is likely to result in variable types of biotic communities across the headwaters of the Cam River.

4.4.2 How spatially structured were ecosystem processes?

Connectivity between sites was important in structuring the spatial pattern of ecosystem processes. Spatial eigenvectors constructed using the MEM method, which incorporated the in-stream distance and the connectivity pattern between sites, consistently explained the most spatial variation, compared with the other methods tested. This suggested that the flow of solutes, organic matter, sediments and propagules of algae or microbes between sites were important in determining the spatial patterns in each measure of ecosystem processes.

Some spatial variation in all measures of ecosystem processes was modelled by eigenvectors with small values which describe local structures (Borcard and Legendre 2002). This is likely due to the influence of sites directly connected by flow, which are thus under very similar water quality conditions and potentially the same land-use, riparian and in-stream conditions. However, with all ecosystem process measures, significant spatial vectors also corresponded to the larger scales which describe global structures (Borcard and Legendre 2002). This indicated that the distance and upstream-downstream flow connectivity between sites explained variability in ecosystem processes at multiple spatial scales. The gap in the range of spatial predictors at the small to medium scale may indicate that the entry of tributaries disrupts the spatial pattern. At larger scales, the impact of confluences is not felt, potentially buffered-out by the volume of water in the main stem, or its overriding landscape or in-stream conditions. The composite measure of ecosystem processes is spatially structured across all scales, as it takes into account the spatial structuring of each type of

ecosystem process measured, which have different absolute spatial patterns and levels of autocorrelation. Composite measures of ecosystem processes are more likely to reflect broad scale spatial and environmental influences, and should be considered for use when attempting to manage system wide stream health.

4.4.3 Spatial versus environmental control of ecosystem processes

The variation partitioning technique I used in this study has been used to identify the various combinations of spatial and environmental controls on community structure and then related to the metacommunity processes that contribute to variation in species assemblages (Cottenie 2005, Peres-Neto and Legendre 2010). These processes can be simplified into a gradient between: neutral or patch dynamics models (communities structured primarily by species loss or dispersal) which can be inferred when spatial structure independent of environmental structure is detected; and niche models (communities structured primarily by the local environment) which can be inferred when communities and environmental structure independent of spatial structures (Leibold et al. 2004, Cottenie 2005). The flexibility of this method means that it can be easily transferred to streams, where the spatial portion can be used to describe processes related to downstream dispersal of organisms (Brind-Amour 2005) and water more generally (Lacey et al. 2007).

I have taken a novel approach and extended the application of variation partitioning to the task of detecting the processes defining variation in ecosystem processes throughout a stream network. Each of the ecosystem processes I investigated is likely to be related to the condition of the water passing through a reach, which through the flowing nature of streams is highly spatially structured, particularly in the Cam River (Chapter 1). However, the processes are also likely to be governed by the in-stream, riparian and land-use conditions at each site, and thus will also be environmentally structured. I attempted to isolate the spatial and environmental contributions to the rates of ecosystem processes by defining the environmental effects at a site, not only locally, but

also by the condition of the environment and the catchment feeding water to that site, and by the condition of the riparian zone along upstream reaches. In this way, spatial effects of water flowing downstream are isolated from the environmental effects that may be conditioning the water on its path downstream to a particular site. The forward selection procedure I used to select significant environmental variables, from the many I defined, took into account the level of inter-correlation between variables when selecting those that best explained the spatial pattern in each measure of ecosystem processes.

All processes studied were highly spatially structured, but the environment (with the exception of the rate of leaf decomposition) and the relative role of space and the environment also varied independently. The large independent effect of spatial structure on algal production, microbial respiration and the DIN:SRP ratio suggests that dispersal of nutrients or water of particular physical form (i.e. high temperature, or turbid) from upstream sites is an important controller of spatial variability in these ecosystem processes. For algae, the downstream dispersal of colonising propagules is also likely to be an important factor in determining the spatial patterns of algal productivity (Swanson and Bachmann 1976). The composite measure of ecosystem processes is also spatially structured, thus the water quality of the many small tributaries of the Cam River has a large impact on the rates of ecosystems processes at down-stream sites, and in turn, the health of sites for the biota that inhabit them.

Space had no independent impact on leaf decomposition rates throughout the Cam River, implying that local conditions are most important for this variable. This is confirmed by the largest portion of variance explained by environmental factors, being for leaf decomposition. Therefore, variation in leaf decomposition rates may be useful for detecting local impacts when this is the goal of a study, yet less useful for system wide monitoring (particularly so when few sites are used). There was a large shared component of variation in each measure of ecosystem processes, where spatial structuring and environmental processes were confounded. In these situations, either or both

could be important but they cannot be statistically separated. The largest amount of unexplained variation was present in the model of spatial and environmental impacts on the composite measure. As much variation in the composite measure of ecosystem processes could not be explained by space or the environment, this may not be the most useful way to study the impacts of land-use intensification and riparian loss on stream health.

The specific environment of each site had a significant influence on microbial respiration, leaf matter decomposition, and the composite measure of ecosystem processes. The environmental fraction was greatest for leaf decomposition. Although these two measures were significantly influenced by the environment, different variables were important within this environmental contribution. This difference in environmental drivers of spatial patterns helps to explain why leaf decomposition and microbial respiration were not correlated as I expected. Size-related variables, stream order and upstream natural stream length are important for leaf matter decomposition, which decreased with increasing stream size (as shown in the Quantile regression); whereas nitrate, and proportion of a catchment in sheep and beef, were important for microbial respiration. As expected, nitrate had a positive relationship with microbial respiration, comparable to the findings of others (Fuss and Smock 1996). The influence of sheep and beef is negative; this may be driven by increased sedimentation caused by stock access to waterways, which has been shown to reduce respiration rates (Hagen et al. 2006, Gulis and Suberkropp 2003). As changes in land-use alter multiple conditions that impact on metabolic rates, many studies reveal, similarly to mine, that multiple interacting factors are responsible for respiration on leaf litter (Niyogi et al. 2003).

Stream order and nitrate were selected for across most models. Although the environmental fractions they contributed to were not always significant, this highlights their influence over ecosystem processes in general. Both were selected as explaining variation in the composite measure of processes, although the environmental fraction (when combined with space) only explained 6% of the variation in the composite measure - this was still significant. A small but

significant link between a stressor such as nitrate and stream health is important, especially in light of the high level of spatial structuring in the Cam River. This information could allow managers to target particular locations in Cam River where nitrate input to the stream should be monitored, especially as much nitrate is contributed by ground water upwelling in this system (Chapter 2). This is another example of the applicability of a spatially explicit approach to targeted within-catchment management.

My adaption of the variance partitioning approach (originally designed for use in community ecology) to identify factors controlling ecosystem processes has proven the flexibility of such methods and should be an encouragement to others in many fields of science to consider their use, especially in situations where spatial correlation exists. One caveat, however, is that care that must be taken with this approach, as the results depend on the quality and relevance of environmental data available, and conclusions must take this into account (Laliberté et al. 2008). It may also overestimate spatial influence, but at least then we can conclude strongly for any environmental effects we find, compared with other studies that make conclusions despite not considering spatial contributions.

The flexibility of spatial eigenfunctions methods has made them applicable to modelling spatial variation in streams. By incorporating such a spatially explicit approach in my analysis, I have been able to successfully tease apart factors driving the processes central to the functioning of streams. In such a complex system as the Cam River it would be difficult and laborious to examine the spatial patterns from a purely observational approach. Overall, my work highlights the importance of protecting headwaters as these are where the extremes occur. Also, due to spatial connectivity within systems such as the Cam River, the water quality and conditions of the headwaters propagate downstream. Rates of ecosystem processes are a more readily understood measure of stream health than water chemistry alone, and may be line with specific end-targets of management, such as to reduce the level of algal from a visual perspective.

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